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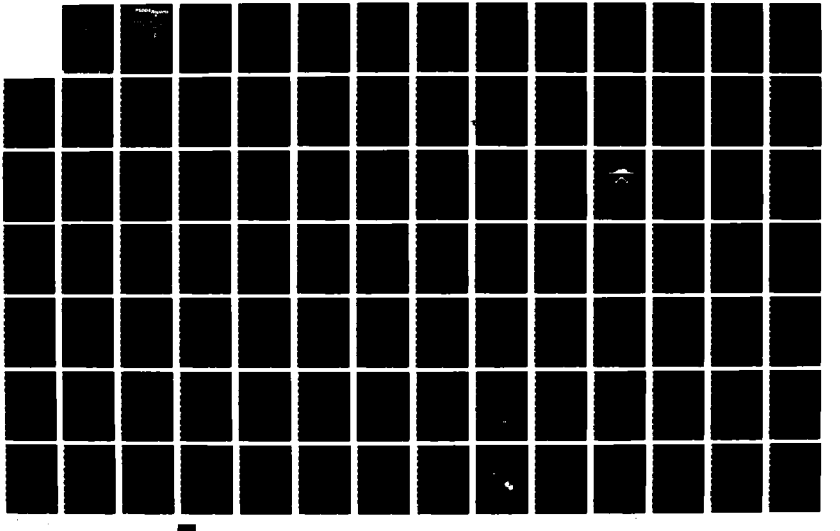

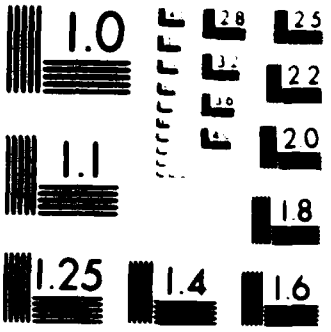
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PSDDA Reports

Puget Sound Dredged Disposal Analysis

Washington State Dept. of Natural Resources

FRAMEWORK FOR COMPARATIVE RISK ANALYSIS OF DREDGED MATERIAL DISPOSAL OPTIONS

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EPA Region 10

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A FRAMEWORK FOR COMPARATIVE RISK ANALYSIS OF DREDGED MATERIAL DISPOSAL OPTIONS

by

Tetra Tech, Inc.

prepared for

Resource Planning Associates

for

Puget Sound Dredged Disposal Analysis
c/o U.S. Army Corps of Engineers
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October, 1986

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CONTENTS

FIGURES

TABLES

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The primary author of this report was Dr. Robert A. Pastorok. Mr. Pieter N. Booth of Tetra Tech contributed to the Executive Summary and the Site Characterizations in the example Comparative Risk Analysis. Dr. Leslie Williams and Mr. Jonathan Shields of Tetra Tech provided the examples of toxicity profiles and spreadsheets for display of risk assessment assumptions and data. Dr. Curtis Brown of the Bureau of Reclamation provided the example of the Range Estimating Program in Appendix D. Dr. Alan M. Ehrlich of U.S. Environmental Protection Agency (EPA) and Dr. Lawrence W. Barnthouse of Oak Ridge National Laboratory provided other example results from uncertainty analysis with risk assessment models. The following reviewers provided valuable comments on the draft report: Dr. Michael Dourson, Dr. Alan Ehrlich, and Mr. Brian Ross of U.S. EPA, Dr. David Jamison of Washington Department of Natural Resources, Mr. Jim Krull of Washington Department of Ecology, Dr. Gary Minton of Resource Planning Associates, and Mr. Keith Phillips, Mr. Michael Scuderi, and Mr. Frank Urabeck of U.S. COE. Ms. Marcy Brooks-McAuliffe and Dr. W. James Erckmann of Tetra Tech assisted in technical editing and report production.

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1985, in Seattle, Washington. The workshop was jointly sponsored by U.S. EPA Region X and PSDDA. Primary participants at the workshop are listed below:

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Dr. Peter Mellinger	Battelle Northwest Laboratory
Dr. Curtis Brown	Bureau of Reclamation
Dr. David Eaton	University of Washington
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Dr. David Rosenblatt	U.S. Army Medical Bioengineering R&D Laboratory
Dr. Lawrence Barnthouse	Oak Ridge National Laboratory
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Mr. Stephen Norsted	Washington Department of Social and Health Services
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Ms. Diane Martin	Envirosphere Company
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Mr. Keith Phillips	U.S. Army Corps of Engineers

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EXECUTIVE SUMMARY

This report was prepared as part of the Evaluation Procedures component of the Puget Sound Dredged Disposal Analysis (PSDDA). The main objective of PSDDA is to provide an environmentally and publicly acceptable management plan for the disposal of dredged material at unconfined sites in Puget Sound. A major feature of the Evaluation Procedures component of PSSDA is to compare the relative risks associated with different disposal options. To this end, the task represented by this report has four objectives:

1) Develop a risk analysis framework for evaluating dredged material disposal options.

2) Develop a hypothetical example of a comparative risk analysis.

3) Develop an example exposure assessment.

4) Develop guidelines for acceptable concentrations of chemical contaminants in marine organisms.

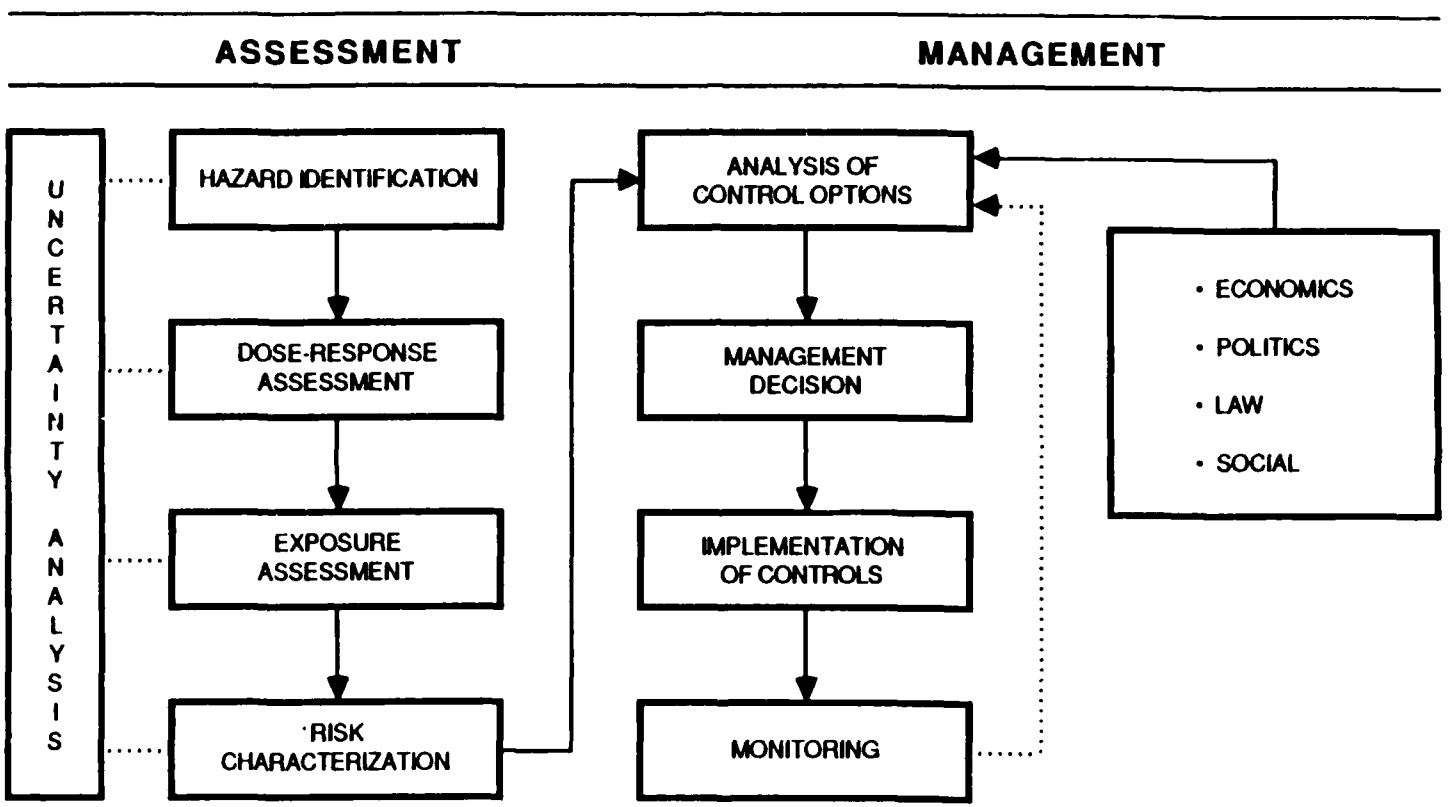
The first three objectives are met by the two main sections of the report: a Conceptual Approach to Risk Analysis of Dredged Material Disposal Options, and an Example of Comparative Risk Analysis. The example is based on a hypothetical, but realistic, disposal scenario wherein the lower Duwamish River system is the source of dredged sediments with three disposal options: 1) Fourmile Rock as a deepwater unconfined site; 2) Piers 90-91 as a nearshore site; and 3) Midway Landfill as an upland site. The last objective is satisfied in the Appendix material.

As part of this project, PSDDA held a workshop on risk analysis for dredged material dispoal on December 16-17, 1985. The main objective of the workshop was to develop work group recommendations for integrating ecological and human health risk information for use in risk management of contaminated sediment. The results of the workshop are synthesized within the context of the risk analysis framework developed in this report. Nevertheless, the approaches suggested herein may not represent a consensus of workshop participants.

CONCEPTUAL APPROACH

Risk is essentially the probability of harm. In the context of this report, risk is the chance (probability) that exposure of humans or biological populations to toxic chemicals will result in a specific adverse effect (e.g., death, liver cancer). The toxic chemicals of concern here are contaminants in dredged material placed at a disposal site. Risk analysis therefore provides a framework for evaluating the potential environmental and human health effects of alternative dredged material disposal options.

As shown in the figure on the following page, risk analysis as defined in this report consists of two processes: risk assessment and risk management. **Risk assessment** is a scientific procedure to determine the probability



Overview of risk analysis.

of adverse effects that may result from a specific exposure to a toxic agent. The results of a risk assessment must be interpreted in light of the estimated accuracy of results, social perceptions of risk, the population at risk, and regulatory requirements. This interpretation and related action is called **risk management**. Risk management translates the scientifically obtained results of risk assessment into a social decision regarding appropriate actions to minimize the risks.

The conceptual approach to risk analysis has six major components (see figure):

- **Hazard identification** - What chemical contaminants are present?
  - What are the potential toxic effects of these chemicals?
- **Dose-response assessment** - What is the relationship between the amount of exposure to a chemical and the prevalence of the toxic effect in a population?
- **Exposure assessment** - What biological resources and human populations are exposed to chemical contaminants?
  - What is the magnitude, duration, and route of exposure(s)?
- **Risk characterization** - What is the probability of toxic effects from the estimated exposure?
- **Comparative analysis** - How do the risks of alternative disposal options compare?
- **Uncertainty analysis** - What is the degree of confidence in the answers to the above questions?

The following discussion is organized by these components.

Hazard Identification

Hazard identification involves defining toxicological hazards posed by individual chemical contaminants in dredged material. Contaminants of concern are selected by listing all identified contaminants, documenting their concentrations, and calculating rank scores to indicate relative toxicities. The rank scores and toxicity profiles are based on consideration of physical-chemical properties such as persistence, organic carbon partition coefficient, and potential carcinogenicity (U.S. EPA 1984).

Dose-Response Assessment

Dose-response assessment is performed for humans based on toxicological indices, and for other species based on sediment bioassays using a series of dilutions of dredged material. The toxicological index used for carcinogens is the **carcinogenic potency factor** which is estimated by the upper 95 percent confidence limit of the slope of a straight line calculated by the linearized

multi-stage dose-response relationship. The toxicological index used for noncarcinogens is the **Reference Dose** (RfD) value which is the maximum average daily exposure that over a lifetime would not be expected to produce adverse effects. For ecological risk assessment, bioassays currently specified for dredged material testing in Puget Sound measure lethal responses in a test of specified duration (e.g., 10 days for the amphipod Rhepoxynius abronius). Additional dose-response relationships may be defined for individual contaminants based on available literature.

Exposure Assessment

Exposure is defined as the contact of an organism with a chemical or physical agent. Exposure assessment involves estimating the magnitude, duration, and route of exposure. Components of exposure assessment include describing the following four elements:

1) Environmental pathways and uptake routes for dredged material or chemical contaminants

2) Concentrations of dredged material or chemical contaminants in various media (air, water, sediments) in space and time

3) Exposed populations and resources

4) Exposure dose (e.g., average rate of intake of a chemical contaminant by an individual of the exposed population).

Each of the elements just listed may vary with specific dredged material and disposal alternatives.

Exposure assessments will vary in complexity depending on the disposal environment, contaminants of concern, transport and fate mechanisms, and the suspected population at risk. Three levels of analyses are discussed, ranging from a qualitative, generic analysis (Level 1); to quantitative exposure estimates (Level 2; estimates expressed as ranges for humans, estimates based on direct measurements for nonhumans); to estimating quantitative changes in exposure over time (Level 3).

Risk Characterization

Risk characterization combines the results of dose-response assessment and exposure assessment to estimate the probability and extent of adverse impacts associated with contaminants in dredged material. This step in the risk analysis can also be performed at three levels of analysis, corresponding to the levels of analysis used in the previous step (Exposure Assessment). Risk characterization is discussed in terms of human health risk and ecological risk.

Human health risk characterizations treat carcinogens and noncarcinogens separately. A plausible upper limit to excess lifetime risk of cancer is calculated using a linearized multi-stage dose-response model and carcinogenic potency factors obtained from U.S. EPA. Excess risk is defined as risk associated with only the disposal site and associated routes of interest (i.e., a marginal increase in risk).

The general model for estimating excess lifetime carcinogenic risk is:

$$PR_{ijk} = B_{ij} E_{ijk} \quad (6)$$

where:

$$E_{ijk} = \frac{C_{ik} I_{jk} X_{ijk}}{W} \quad (7)$$

and:

PR_{ijk} = Excess lifetime risk of cancer from intake of contaminant i via route j and medium k

B_{ij} = Unit risk or carcinogenic potency factor for contaminant i and exposure route j $[(mg \cdot kg^{-1} \cdot day^{-1})^{-1}]$

E_{ijk} = Average lifetime exposure dose of chemical i via route j and medium k $(mg \cdot kg^{-1} \cdot day^{-1})$

C_{ik} = Average concentration of chemical i in medium k (mg/kg)

I_{jk} = Contact rate (e.g., ingestion or inhalation rate) with medium k via route j (kg/day)

X_{ijk} = Absorption coefficient for uptake of chemical i via route j and medium k (dimensionless)

W = Reference human weight = 70 Kg.

An index of noncarcinogenic risk may be approximated as the ratio of the estimated exposure to the Reference Dose (RfD) as follows

$$HI_{ij} = \frac{1}{RfD_{ij}} \sum_{k} E_{ijk} \quad (8)$$

where:

HI_{ij} = Hazard Index, or indicator of intake of chemical i via route j relative to the Reference Dose (RfD) corresponding to a No Observed Adverse Effect Level (NOAEL) (dimensionless)

E_{ijk} = Estimated average lifetime exposure dose of chemical i from medium k via route j $(mg \cdot kg^{-1} \cdot day^{-1})$ calculated as defined earlier

RfD_{ij} = Reference Dose of chemical i via route j $(mg \cdot kg^{-1} \cdot day^{-1})$

Risk estimates used in the final comparative analysis model are net excess risk calculated by subtracting the risk associated with reference area sediment from the risk associated with the dredged material.

For purposes of this report, ecological risk characterizations will differ from human health risk characterizations primarily in that effects are measured in terms of mortality rather than sublethal responses. Methods are illustrated for estimating area-specific risks to migrant and nonmigrant individuals (Level 2 analysis). For example, the model used to estimate area-specific risk to an individual of a nonhuman species based on Level 2 exposure analysis is:

$$PM_{xr} = PE_{xr} \cdot PB_{xr}[C(x)] \qquad (12)$$

where:

PM_{xr} = Probability (risk) of mortality of an individual of resource species r due to exposure at location (area) x

PE_{xr} = Probability of exposure of an individual of species r at location x, as a function of resource overlap with location x(O_{xr}) and area occupied by resource r (A_r).

$PB_{xr}[C(x)]$ = Probability of mortality of an individual of species r upon exposure to concentration, C(x), of dredged material at location x. The relationship between probability of mortality and concentration is based on a dose-response curve from a series of sediment bioassays

C(x) = Concentration (or percentage) of dredged material at location x.

To account for burial of sensitive species, the probability of mortality (PM_{xr}) of sensitive organisms is set equal to 1.0 whenever the thickness of the sediment deposit exceeds a specified threshold.

Four different approaches are also described for adding temporal variability to the estimate of ecological risk (Level 3 analysis). These approaches are: 1) time-averaged exposure and risk; 2) frequency of unacceptable exposure; 3) time variable uptake and depuration kinetics; and 4) population modeling.

Comparative Analysis Model

Workshop participants recommended using the Multi-Attribute Tradeoff System (MATS) model described by Brown and Valenti (1983) for integrating ecological and human health risk estimates to evaluate various options for dredged material disposal. The MATS Model allows a user to place relative values on a variety of attributes (e.g., risk estimates or cost-effectiveness measures for disposal options) that are not directly comparable. For example, risks to terrestrial wildlife from land disposal are not directly comparable to risks to fish from aquatic disposal. Tradeoffs between risks to land and water resources must be considered when evaluating land vs. aquatic disposal options. Once relative values are placed on terrestrial wildlife and fish, for example, the decision-maker can rank disposal options by using MATS to integrate rank scores for the various attributes.

Key steps in conducting the MATS evaluation are selecting key attributes, creating numerical scales (comparative-risk scales) and functions for each attribute to allow comparison of attributes with different original units, and assigning weights to each attribute. Key attributes (or variables) selected for dredged material disposal include cost, sediment chemistry, bioaccumulation, toxicity bioassay, and human health risk. The MATS model allows the analyst to incorporate shifts in perceived risk based on professional judgment, agency policy, or public perception. Finally, the analyst performs an evaluation of disposal options using MATS by summing weighted risk indices for each option.

Uncertainty Analysis

Risk assessments are always based on limited data, analytical assumptions, and models that by definition are imperfect. It is therefore essential to discuss the uncertainties associated with estimates of exposure, toxicological hazard, and risk. Three examples of uncertainty analysis are discussed: order-of-magnitude bounding analysis, probability distributions for risk, and model uncertainty.

Two examples of order-of-magnitude bounding analysis are the Range Estimating Program discussed at the workshop by Dr. Curtis Brown, and results of U.S. EPA work provided by Dr. Alan Ehrlich. The Range Estimating Program uses information on means and ranges of key analysis variables and performs a Monte Carlo simulation to estimate probability distributions of outcomes. The bounding analysis results described by Dr. Alan Ehrlich illustrated the use of ranges of values for model coefficients to calculate the resultant uncertainty range of risk estimates.

Probability distributions of human health risk were illustrated by Crouch et al. (1983), who modeled uncertainty in carcinogenic potency, exposure, and an interspecies extrapolation factor as lognormal probability distributions. This analysis yielded: 1) a lognormal probability distribution for excess cancer risk, 2) median, mean, and 98th percentile estimates of risk, and 3) the variance of the mean of lognormal transformation of risk.

Uncertainty in models for extrapolation from high doses used in bioassays to low doses of interest has been exhaustively investigated by risk analysts. Given the wide range of risk estimates derived from dose-extrapolation models, choice of a model must be based on best scientific judgment of biological realism and best judgement of policy direction. Both model uncertainty and parameter uncertainty may be investigated by qualitatively examining the assumptions of the model.

EXAMPLE APPLICATION OF COMPARATIVE RISK ANALYSIS

In the second section of this report, the conceptual approach to risk analysis is applied to a hypothetical dredged material disposal scenario. The Fourmile Rock Disposal Site, a deepwater, unconfined site is analyzed in terms of site characterization, hazard identification, dose-response assessment, exposure assessment, and risk characterization. Comparative analysis of disposal options is illustrated using the MATS model to evaluate

the risks determined for the deepwater site relative to hypothetical risks for nearshore confined disposal and upland disposal.

The risk analysis described in this report can be undertaken at several levels of detail, and can be adapted to the needs and resources of a range of projects. This analysis may be used as a flexible management tool for identifying critical tests that will have the greatest influence on disposal decisions for dredged materials. By identifying important concerns through the risk analysis process, available resources may be allocated effectively. The risk analysis framework presented in this report allows the flexibility to select a specific modeling approach that is appropriate for a particular problem. Actual applications of risk analysis to specific cases of dredged material disposal will be needed to refine the approach and evaluate alternative models.

INTRODUCTION

This report was prepared in support of the Puget Sound Dredged Disposal Analysis (PSDDA). The comparative risk analysis was performed as Task 5b of an overall effort to develop Sediment Quality Values for Puget Sound (Tetra Tech 1986a), but the final report for this component has been produced separately. The main objective of PSDDA is to provide an environmentally and publicly acceptable management plan for the disposal of dredged material at unconfined sites in Puget Sound. A key element of the Evaluation Procedures component of PSSDA is to consider the relative risks associated with the different disposal options. As specified in the Work Plan (August 1985), the objectives of Task 5b are to:

- Summarize a consensus technique for assessing site-specific risks associated with different disposal sites as recommended by a workshop of experts
- Tabulate the health risk associated with the consumption of contaminated tissues.

These objectives were expanded and clarified after the workshop of experts was held in December 1985 to include the following elements:

- Develop a risk analysis framework for evaluation of dredged material disposal options (e.g., upland vs. nearshore vs. deepwater unconfined disposal sites) relative to human health and environmental impacts
- Present a hypothetical example of a comparative risk analysis for upland, nearshore, and deepwater unconfined sites using the Multi-Attribute Tradeoff System (MATS) model
- Provide an example exposure assessment with quantitative estimates for both human health and ecological risk components
- Develop guidelines on acceptable concentrations of chemical contaminants in marine organisms based on health risk models, assuming a range of seafood consumption rates for a hypothetical human population.

The risk analysis framework and guidelines developed in this report may be used with other guidelines, such as Sediment Quality Values developed for PSDDA and the Puget Sound Estuary Program (Tetra Tech 1986a), to determine the relative suitability of alternative disposal sites for dredged material in the Puget Sound Basin. As such, this work may provide a key conceptual element of the Evaluation Procedures component of PSDDA.

PSDDA will provide the basis for publicly and environmentally acceptable disposal plans and guidelines for unconfined, aquatic disposal of dredged

material into Puget Sound. The objectives of PSDDA are to 1) locate acceptable sites for open-water unconfined disposal of dredged material in Puget Sound, 2) identify chemical and biological evaluation procedures for determining the acceptability of dredged material for specified disposal options, and 3) develop management plans for dredged material disposal sites. Although the focus of PSDDA is on unconfined, open-water aquatic disposal of dredged material, the evaluation procedures developed under Objective 2 above will address alternative methods of disposal, including upland, nearshore confined, offshore confined (e.g., capping), and ocean disposal options. Specific disposal sites will be selected and evaluated for the unconfined, open-water option only. The alternative disposal options just described will be assessed on a generic basis.

Risk analysis provides a framework for evaluating the potential environmental and human health effects of alternative dredged material disposal options. Under PSDDA, risk analysis may initially be applied to help decide what level of dredged material contamination is acceptable for unconfined, open-water disposal. Risk analysis is especially appropriate for dealing with complex environmental problems associated with potential exposures to toxic chemicals (National Research Council 1983; Rodricks and Tardiff 1984; Ricci 1985).

The primary purpose of risk analysis is to estimate the probability of occurrence of a specified environmental effect associated with toxic chemical exposures. Effects may include ecological impacts (e.g., risk of mortality or reproductive failure in fish populations) or human health effects (e.g., risk of cancer associated with consumption of contaminated seafood).

As part of this project, PSDDA held a workshop on risk analysis of dredged material disposal on December 16-17, 1985. The main objective of the workshop was to develop work group recommendations for integrating ecological and human health risk information for use in risk management of contaminated sediment. Secondary objectives included development of conceptual frameworks for exposure assessment and uncertainty analysis. The workshop consisted of presentations on various aspects of risk analysis, discussions of issues among workshop participants, and consensus-building on approaches to risk analysis. The workshop agenda is provided in Appendix B. The results of the workshop are synthesized within the context of the risk analysis framework developed in this report. Nevertheless, the approaches suggested herein may not represent a consensus of workshop participants.

The remainder of this report is organized into two major sections:

- Conceptual Approach to Risk Analysis of Dredged Material Disposal Options
- Example Application of Comparative Risk Analysis.

In each major section, the steps of the risk analysis method are illustrated. An example exposure assessment is presented within the second chapter listed above. Complete specification of exposure assessment models (e.g., equations) for all disposal options was beyond the scope of this work. However, a discussion of available models and guidance on important variables and processes to be modeled is provided.

Several other factors guided the development of the risk analysis framework presented below. First, the conceptual approach for this risk analysis was designed to complement present evaluation procedures and dredged material testing strategies [U.S. Army Corps of Engineers (COE) 1985]. Second, it was assumed that the risk analysis had to be a predictive assessment. Direct monitoring of human exposures is impractical for the low doses of interest here. Although monitoring of ecological variables during and after disposal operations is one component of dredged material management, monitoring data specific to a project would not be available for initial assessments. Third, the risk of failure in site-containment designs (e.g., leakage through walls constructed around a nearshore or an upland site) was not considered. Specific site designs were assumed to be constant. Finally, each risk variable is defined as a measure of excess risk. Only risks associated with a specific disposal option would be estimated during application of the models. For example, risks of cancer in a human population due to environmental contamination from an industrial facility near the dredged material disposal site would not be assessed. In every instance, the measure of interest is the **marginal increase in risk** due to the disposal of contaminated sediments in a specific environment.

CONCEPTUAL APPROACH TO RISK ANALYSIS OF DREDGED MATERIAL DISPOSAL OPTIONS

Risk analysis as defined in this report consists of two processes: risk assessment and risk management (Figure 1). Risk assessment is the scientific determination of the probability of adverse effects resulting from a specific exposure to toxic agents. Risk assessment includes one or more of the following:

- Hazard identification
- Dose-response assessment
- Exposure assessment
- Risk characterization.

An outline of risk analysis for evaluation of dredged material disposal options is given in Table 1. Because uncertainty is pervasive in risk analysis problems, uncertainty analysis is an important element of each stage of the assessment process.

Risk management is the translation of scientific information into public policy and action to control risks. In essence, the results of a risk assessment are evaluated in the context of regulatory legislation and socioeconomic, technical, and political factors to decide on a "socially acceptable" level of exposure to suspected toxic agents. Risk management includes four principal stages:

- Analysis of control (risk reduction) options or alternative actions such as dredged material disposal options
- Management decision (e.g., selection of a preferred option)
- Implementation of selected control option or action
- Monitoring.

The scheme in Figure 1 accounts for the fact that agency policy and public perception influence management decisions about the social acceptability of predicted risks in specific circumstances.

Following a risk assessment, a management decision must consider the relative and absolute risks for various options, as well as economic, political, legal, and social factors. Another key consideration in managing risk is to integrate diverse forms of information into a model or conceptual framework that provides a net estimate of risk or a preference ranking for alternative actions. For example, during the course of risk assessment/management for dredged material disposal options, several tests may be conducted on dredged material, yielding a series of risk estimates for

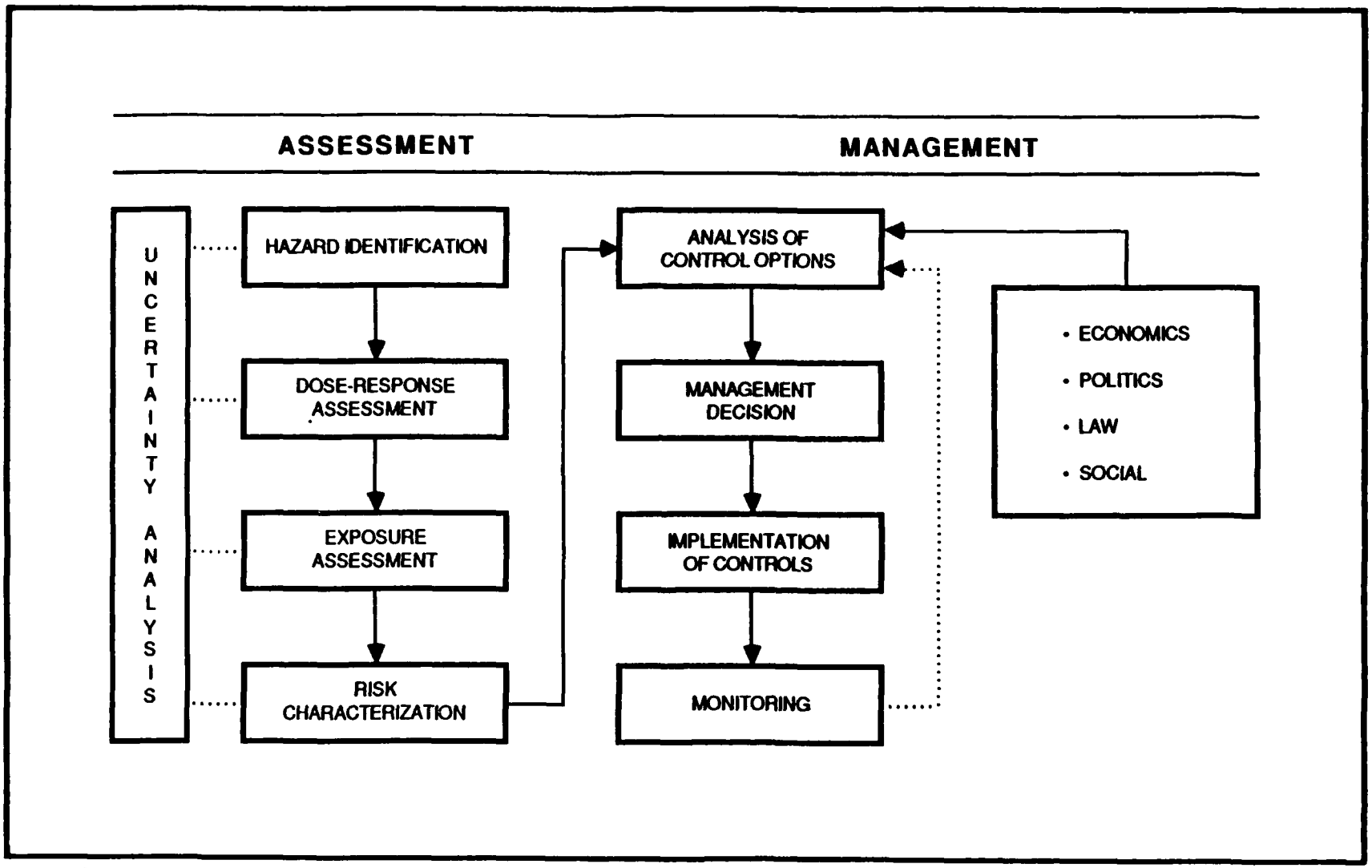


Figure 1. Overview of risk analysis.

TABLE 1. STEPS IN RISK ANALYSIS OF DREDGED MATERIAL DISPOSAL

I. HAZARD IDENTIFICATION

- o Identify chemicals of concern
- o For each chemical of concern, characterize
  - Physical/chemical properties
  - Metabolic and pharmacokinetic properties
  - Toxicologic effects
  - Structure activity relationships (if necessary)
- o Evaluate uncertainty

II. DOSE-RESPONSE ASSESSMENT

- o Compile toxicologic indices; e.g., ADIs and carcinogenic potency factors for human effects
- o Summarize results of sediment testing for nonhuman biological effects
- o Evaluate uncertainty

III. EXPOSURE ASSESSMENT

- o Select exposure scenarios (pathways and routes)
- o Estimate environmental concentrations of dredged material (or component chemicals) using transport and fate model
- o Inventory economic and recreational resources, potentially exposed human populations, and their proximity to the disposal site(s)
- o Estimate exposure dose and duration
- o Evaluate uncertainty

IV. RISK CHARACTERIZATION

- o Estimate risk for human and non-human populations
- o Summarize uncertainty analysis

TABLE 1. (Continued)

V. RISK MANAGEMENT (COMPARATIVE ANALYSIS)

- Select variables for use in comparative risk analysis matrix
- Select comparative analysis model for evaluating disposal options according to relative ecological and human health risk
- Rank disposal options

Optional

- Define acceptable risk levels
- Eliminate options that result in unacceptable risk for any single variable
- Rank acceptable disposal options

humans and other species. Furthermore, some test results give only an indirect measure of risk with varying units among tests.

The following sections describe hazard identification, dose-response assessment, exposure assessment, risk characterization, comparative analysis, and uncertainty analysis as the major steps in performing a risk analysis of dredged material disposal. Although past authors have not consistently applied a single term to all aspects of this procedure, the phrase "risk analysis" will be used throughout this report to encompass both risk assessment and risk management (e.g., Figure 1). Other terms and definitions used in this report are generally consistent with those provided by National Research Council (1983) and U.S. EPA (1986a,b,c; 1986d). Although National Research Council (1983) includes comparative analysis and uncertainty analysis in risk characterization, these processes are described in separate sections below to emphasize their importance.

HAZARD IDENTIFICATION

The first step in the risk assessment process is to define the particular toxicological hazards posed by individual chemical contaminants present in the dredged material. This step is important because the nature and degree of the hazard will largely determine the extent of subsequent analyses (e.g., exposure assessment and risk characterization). Toxicological hazards are defined by selecting contaminants of concern and constructing toxicity profiles for those contaminants. Toxicity profiles should be compiled from available information and displayed as concise summary tables.

Selection of Contaminants of Concern

Selection of contaminants for consideration in hazard assessment is based partly on the results of analytical chemistry of dredged materials and other available data on sources and contamination at the dredging project site. U.S. COE (1985) lists appropriate documents and records to be checked for data on site contamination.

Guidelines for selection of chemicals of concern for dredged material are discussed by Tetra Tech (1986a). Additional criteria are specified in an initial screening process described in the Superfund Public Health Evaluation Manual (ICF 1985). The process involves four steps:

- List all identified contaminants in dredged material or at the dredging site
- Document environmental concentrations of contaminants in dredged material or at the dredging site and mathematically determine representative values (e.g., arithmetic or geometric mean concentrations)
- Calculate rank score based on the method described in ICF (1985), Superfund Public Health Evaluation Manual
- Select chemicals of concern based on rank score and other factors.

The rank score for each chemical is based on the product of the chemical concentration and a toxicity constant. The toxicity constant is derived from minimum effective doses for chronic effects and a factor expressing the severity of effect. Values of toxicity constants and methods of derivation are given in Appendices C and D, respectively, of ICF (1985).

Other factors used for selecting chemicals of concern are related to exposure potential or weight-of-evidence classification for potential carcinogens. Exposure potential criteria include vapor pressure, Henry's law constant, organic carbon partition coefficient, and the chemical's persistence in various media. The weight-of-evidence classification for potential carcinogens (U.S. EPA 1986a) dictates that chemicals with sufficient evidence of carcinogenicity [i.e., U.S. EPA (1986a) Group A or International Agency for Research on Cancer (IARC 1982) Group 1 chemicals] should generally be designated as contaminants of concern regardless of other factors.

Toxicity Profiles

Toxicity profiles are constructed for the selected chemicals of concern by summarizing the following information:

- Physical-chemical properties (e.g., vapor pressure, octanol-water partition coefficients)
- Metabolic and pharmacokinetic properties (e.g., metabolic degradation products, depuration kinetics)
- Toxicological effects for specific uptake routes (e.g., target organs, cytotoxicity, carcinogenicity, mutagenicity).

Toxicity profiles prepared specifically for use at hazardous waste sites are available for approximately 195 chemicals from U.S. EPA (Office of Waste Programs Enforcement and Office of Environmental Criteria and Assessment). Additional information on physical-chemical properties may be obtained from computerized databases (e.g., Chemical Information System) or specialized references (e.g., Lyman et al. 1982; Callahan et al. 1979).

Toxicity profile data are used to characterize the key toxicologic properties of contaminants of concern. Moreover, these data influence the nature and extent of subsequent steps in risk analysis. For example, the endpoint of concern in dose-response assessment may be selected based on the most severe adverse effect identified in the toxicity profile. When data are inadequate for a quantitative risk determination, the toxicity profile serves as the product of the risk assessment. Thus, U.S. EPA (1986a) refers to hazard identification as qualitative risk assessment.

DOSE-RESPONSE ASSESSMENT

After characterizing the potential hazard associated with individual chemical components of the dredged material, the relationship between dose of a substance and its biological effect is determined. Dose-response relationships may be developed for individual contaminants and for bulk sediments (chemical mixtures). In general, data for the former would not be generated for individual projects, but would be determined from the

literature. In conducting dredged material risk assessments, two types of dose-response information are used: human health dose-response, and dose-response for nonhuman species.

Human Health

Dose-response assessment for human health risk determinations are based on toxicologic indices for two broad categories of toxicants as follows:

- **Carcinogens** are each characterized by a **carcinogenic potency factor**, which is a measure of the cancer-causing potential of a substance. A plausible upper bound for the carcinogenic potency factor is represented by the upper 95 percent confidence limit of the slope of a linear dose-response curve, usually derived from the linearized multi-stage model (U.S. EPA 1986a)

- **Noncarcinogens** are each characterized by an **Reference Dose (RfD) Value.** The RfD is the average daily exposure, which over a lifetime would not be expected to produce adverse effects.

The lack of a demonstrated threshold in dose-response relationships for carcinogens (U.S. EPA 1980b, 1986a; U.S. Office of Science and Technology Policy 1985) implies a finite risk of cancer even at very low doses of the carcinogen. Therefore, a quantitative risk assessment approach is used to predict an upper limit estimate of the probability (risk) that a given exposure level will result in cancer. The potency of a carcinogen (i.e., the carcinogenic potency factor) is a plausible upper limit estimate of the probability of effect per unit dose of chemical in units of $(mg \cdot kg^{-1} \cdot day^{-1})^{-1}$ or $kg \cdot day \cdot mg^{-1}$ (i.e., the inverse of exposure units, $mg \cdot kg^{-1} \cdot day^{-1}$). For a noncarcinogen, an acceptable exposure value (the Reference Dose, or RfD in units of $mg \cdot kg^{-1} \cdot day^{-1}$) is defined based on a threshold dose below which no adverse biological effects are expected.

Carcinogenic potency factors and RfDs for U.S. EPA priority pollutants are provided in Appendix C. Methods and data used to derive these values are given by U.S. EPA (1980b, 1986a, 1985a) and Stara et al. (1983). Note that conservative assumptions are made by U.S. EPA in deriving these indices so that final risk estimates based on them will be protective of human health. Other sources of RfDs are listed by Rosenblatt et al. (1983).

Nonhuman Species

Dose-response relationships for nonhuman species are based on:

- Sediment bioassay tests (U.S. COE 1985) using bulk sediment or some fraction thereof (e.g., elutriate, effluent from initial disposal operation)

- Data on toxicity tests for individual contaminants, media, and species of concern.

For each species, medium, and chemical (or bulk sediment), data should be plotted as the percent of test population exhibiting the response vs. the concentration of substance. A family of curves may be plotted if tests of different durations are conducted. At present, bioassays specified for dredged material testing in Puget Sound measure acute lethal responses in a test of specified duration (e.g., 10 days for amphipod Rhepoxynius abronius sediment toxicity bioassay).

More complex dose-response assessments could include a battery of laboratory and field tests at the level of individual species, populations, or communities. A comprehensive review of possible tests is beyond the scope of this report, but the interested reader may consult Gentile et al. (1983a,b), Bierman et al. (1986), and others. One particularly promising approach is life-cycle testing for characterizing changes in age-specific fecundity and mortality patterns, intrinsic rate of population growth, and reproductive value in response to contaminant exposure (Gentile et al. 1982, 1983b).

EXPOSURE ASSESSMENT

Exposure assessment is the process of characterizing the populations exposed to the chemicals of concern, the environmental transport and fate pathways, and the magnitude and duration of the exposure dose. Separate exposure assessments are conducted for human and nonhuman populations. Factors considered in an exposure assessment may include:

- Exposure medium (air, water, soil, biota)
- Environmental transport and fate of contaminants
- Monitored or projected contaminant concentrations at exposure points
- Characterization of exposed populations, including sensitive subpopulations
- Potential routes of exposure (inhalation, dermal contact, oral ingestion)
- Measurement or estimation of numerical parameters (e.g., contaminant contact rates, route-specific absorption rates).

An exposure assessment is performed in three stages. First, an estimate is made of environmental concentrations. This estimate is based on actual concentrations of the chemicals in the dredged material and transport and fate mechanism affecting both the sediments and the chemicals themselves. Second, an analysis is performed to describe the exposed populations. Third, information on estimated concentrations and exposed populations is combined in an integrated analysis to construct exposure profiles for each pathway and route. In the case of dredged material disposal, the results of the exposure assessment depend on the alternative disposal method chosen. For this reason, the different exposure scenarios are discussed first, followed by a discussion of the three stages of the exposure assessment.

Exposure Scenarios

Potential exposure pathways and routes are postulated to characterize mechanisms for transfer of a contaminant in the sediments at a disposal site to a target population or individual. Development of exposure scenarios involves the identification of important transport and fate processes. These processes are often modeled to estimate environmental concentrations of contaminants in later stages of the exposure assessment.

Potential exposure pathways and uptake routes for contaminants in dredged material are summarized in Table 2. A qualitative evaluation of the relative importance of each environmental pathway is given in the table. This evaluation only applies to comparisons within disposal option categories. Comparisons among different disposal options are made during the comparative analysis stage of the risk analysis (see below, Comparative Analysis Model). For example, at this stage of the analysis, the relative importance of water column transport from an unconfined deepwater site is not compared with leachate/groundwater transport from an upland site. Also, no attempt was made to evaluate the relative importance of pathways involving human health concerns vs. those leading to ecological effects. In most cases, however, it is assumed that public policy would dictate a higher concern for human health risks than for ecological effects, at least when similar pathways are involved.

Only the most likely pathways of concern for each environmental media are shown in Table 2. The pathways and routes listed in the table are discussed below for each disposal option.

Unconfined Deepwater Disposal--

- Water-column transport of suspended sediment or dissolved contaminants originating either directly from the disposal plume or from resuspended material

  -- Contaminants may be taken up by marine organisms or humans through dermal absorption or ingestion. If material is transported to the shore, human contact may occur. The importance of this pathway for both ecological effects and human health risks is judged to be low because of the relatively short persistence of the disposal plume in the water column, the low probability of large-scale resuspension of sediments at well-designed deepwater sites, and the expected rapid decline of contaminant concentrations with distance from the disposal site.

- Water-column transport of suspended sediments with subsequent deposition on the shore

  -- Contaminant uptake by organisms may occur through dermal contact or by ingestion of beach sediments. Contaminant uptake by humans will occur predominantly through dermal contact. This exposure pathway is considered relatively unimportant for reasons given under the previous item.

TABLE 2. POTENTIAL EXPOSURE PATHWAYS AND ROUTES FOR EVALUATION OF DREDGED MATERIAL DISPOSAL

Disposal Option	Environmental Pathway	Relative Importance of Pathway[a]	Chemical Uptake Route[a]	Concern
Unconfined, deepwater	Water-column transport	+	D, I	Ecological effects
		+	D, I	Human health
	Sediment contact, on-shore	+	D, I	Ecological effects
		+	D, I	Human health
	Sediment contact, subtidal	+++	D, I	Ecological effects
		+	D	Human health
	Food chain transfer	+++	I	Ecological effects
		+++	I	Human health
Upland or nearshore	Atmospheric transport	+	N, I	Ecological effects
		+	N, I	Human health
	Sediment contact on-site	++	D, I	Ecological effects
		+	D, I	Human health
	Effluent/water transport	++	D, I	Ecological effects
		+	D, I	Human health
	Surface runoff/water transport	++	D, I	Ecological effects
		++	D, I	Human health
	Food chain transfer	+++	I	Ecological effects
		+++	I	Human health
	Leachate/groundwater transport	+++	D, I	Human health
		+++	D, I	Ecological effects

[a] Relative importance ranking only applies to comparisons within disposal option categories. Also, the relative importance rankings are indicated separately for ecological effects and human health (see text).

\+ = Low importance

++ = Moderate importance

+++ = High importance

D = Dermal

I = Ingestion

N = Inhalation.

- Contact with sediment at the disposal site or off-site

  -- This is considered the most important pathway for ecological exposure because contaminant concentrations are highest at the disposal site. Human exposure through this pathway is considered negligible, assuming no direct contact of humans with sediments at the disposal site.

- Food chain transfer of contaminants

  -- Uptake of contaminants and transfer through the food chain is considered relatively important for both marine and human populations. Humans may harvest and consume higher trophic level predators (e.g., predatory fishes) that have accumulated contaminants.

Upland or Nearshore Disposal--

- Atmospheric release and transport

  -- Airborne particles resuspended by wind may be inhaled or ingested by terrestrial wildlife or humans. Atmospheric release and transport of volatile compounds leads to poential exposure of local wildlife and humans via inhalation. The relative importance of these pathways is low because of the low probability of releases at well-designed sites and the relatively low concentrations of contaminants expected in air containing resuspended particles or volatile compounds.

- Contact with sediment at the disposal site

  -- Direct contact with sediment on-site is not a primary pathway for human exposure. Access to the disposal site would always be limited. Ecological concerns may vary greatly depending on species group. For example, contaminants bound to particles may not be very available for absorptive uptake by plants, but at least some chemicals would be taken up efficiently by animals ingesting the sediment.

- Effluent release with subsequent water transport

  -- Release of effluents at the site would be short-lived. Although this pathway is considered potentially more important than the previous one for ecological effects, human health concerns are low. It is assumed that design of the effluent-release system would preclude human exposure to effluent under most circumstances.

- Surface runoff with subsequent water transport

  -- This pathway is considered of moderate relative importance for human health and ecological effects. Drinking of runoff water or dermal contact is possible, but proper site design would preclude significant runoff in most cases. Surface runoff is considered a less important pathway than leachate and groundwater transport because: 1) contaminant mobility is low under oxidized conditions at the sediment surface, 2) infiltration is likely to exceed runoff, and 3) surface water systems are less likely to be used by humans as sources of drinking water than are groundwater systems.

- Food chain transfer of contaminants

  -- Uptake by plants or soil organisms may eventually lead to transfer of contaminants to wildlife or humans. This pathway is assigned a high relative importance. Although proper design of disposal sites should minimize access to both humans and wildlife, contaminants could migrate offsite via groundwater. Subsequent uptake by plants or animals could pose a potential health risk, especially if human croplands or rangeland is contaminated.

- Contamination of leachate and transport into groundwater systems

  -- This pathway is considered especially important because of the potential for contamination of potable aquifers. Uptake of contaminants from soil porewater by nonhuman species is also a likely exposure pathway.

The pathways and processes particular to each disposal site alternative must be taken into account when estimating environmental concentrations--the next step in exposure assessment.

Estimation of Environmental Concentrations

The estimation of exposure concentrations usually involves two stages of analysis:

- Transport and fate analysis of dredged sediment during and after disposal

- Transport and fate analysis of individual contaminants or classes of contaminants during and after disposal.

Although simple transport and fate models may be adequate for estimating contaminant concentrations for time scales on the order of hours or days, long-term predictions may be inaccurate, especially for labile (i.e., unstable) chemicals. In such cases, there is a need for more complex models that

treat individual contaminants separately. Modeling approaches for the transport and fate of sediments and chemicals are described below.

Transport and Fate of Sediments--

Because transport and fate processes differ considerably among disposal scenarios, marine disposal and nearshore/upland disposal are discussed separately.

Marine Disposal--Various mathematical models have been used to predict the initial dispersion and bottom distribution of dredged material discharged from a barge into marine waters (Koh and Chang 1973; Brandsma and Divoky 1976; Johnson 1980; U.S. COE 1985). In general, these models do not address long-term processes such as resuspension, transport, and redeposition of dredged material. Johnson (1980) reviewed several models of the short-term fate of dredged material disposal in the marine environment. Only short-term (hours to days) dispersion and deposition modeling is considered herein.

Discharge of dredged material into Puget Sound is usually accomplished by split-hull barges. Such barges release material rapidly (in times less than 1 min); hence, from a modeling perspective, the release can be considered instantaneous. The first well-known computer model capable of calculating the settling of an instantaneous discharge was written by Koh and Chang (1973) and modified by Brandsma and Divoky (1976). The latter model has been used, with some improvements, by a number of subsequent investigators and has recently been applied to dumping operations by Johnson (3 March 1986, personal communication). This model describes the dynamic descent phase of dredged material disposal more completely than do other models. Therefore, the model of Brandsma and Divoky (1976) as modified by Johnson (3 March 1986, personal communication) is recommended for predicting the fate of barge-dumped sediments in Puget Sound.

Nearshore or Upland Disposal--Release and subsequent transport of dredged material from an upland or nearshore site can occur in three ways:

- Wind erosion and atmospheric transport
- Erosion by surface runoff and surface water transport
- Release of suspended solids in effluent.

Wind erosion rates depend on many factors including wind velocity and direction, physical composition of soil, topography, and ground cover. Geraphty and and Ricci (1985) discuss models for wind erosion and atmospheric transport. Because this is considered a minor exposure pathway for dredged material disposal sites, such models are not addressed further.

Modeling of sediment fate associated with surface runoff or release of effluents can be conducted using approaches discussed below (see Transport and Fate of Chemical Contaminants). Essentially, suspended sediment is treated as a mass-transport variable analogous to a conservative chemical.

Transport and Fate of Chemical Contaminants--

The objective of the analysis of transport and fate of chemicals is to estimate exposure concentrations in various media (air, soil or sediment, water). As discussed below, the analysis may or may not include modeling of time-variable processes such as sediment bioturbation and biodegradation of chemicals. Assessment of environmental transport and fate may include characterizing the following processes or variables:

- **Transport** - volatilization, sorption, precipitation-dissolution, leaching, advection, diffusion, and sedimentation
- **Transformation** - photolysis, hydrolysis, reduction-oxidation, and microbial degradation
- **Bioaccumulation** - empirically determined tissue residues from laboratory bioaccumulation tests on dredged material, substance-specific bioconcentration factors, and octanol-water partition coefficients for nonionic organic contaminants
- **Site-specific conditions** - hydrological, geological, and meteorological conditions that may affect chemical migration.

The extent of the transport and fate analysis depends on the results of sediment testing and possibly the initial mixing-zone calculations (U.S. COE 1985). An example of a decision-making approach to determine the level of analysis is shown in Figure 2. Decision criteria specified in Figure 2 are preliminary and subject to revision upon further review. The first step might be to compare concentrations of contaminants in dredged material with reference values. If contaminant concentrations in dredged material are all less than these reference values, then only a qualitative (Level 1) analysis of transport and fate would be performed. The reference values are not necessarily indicative of "acceptable" material for any given disposal option. Reference values could be derived from sediment quality values described by Tetra Tech (1986a). Similarly, X, Y, and Z values in Figure 2 are guidelines for comparison with dredged material characteristics or variables derived from mixing zone calculations. If conditions predicted for a disposal option exceed the guidelines, then the most intensive (Level 3) analysis of chemical transport and fate would be performed. Mixing zone calculations and the three possible levels of analysis are discussed in the following sections.

Mixing Zone Calculations--Determination of mixing zone dimensions may be required to decide between Level 2 and Level 3 analysis of transport and fate. A mixing zone is defined as the water parcel required to dilute concentrations of contaminants associated with a discharge to some specified level. For example, the mixing zone necessary to achieve dilution of a dissolved chemical of concern to the concentration equal to the water quality criterion depends on:

- Concentration of the chemical in standard elutriate
- Concentration of the chemical in disposal site water

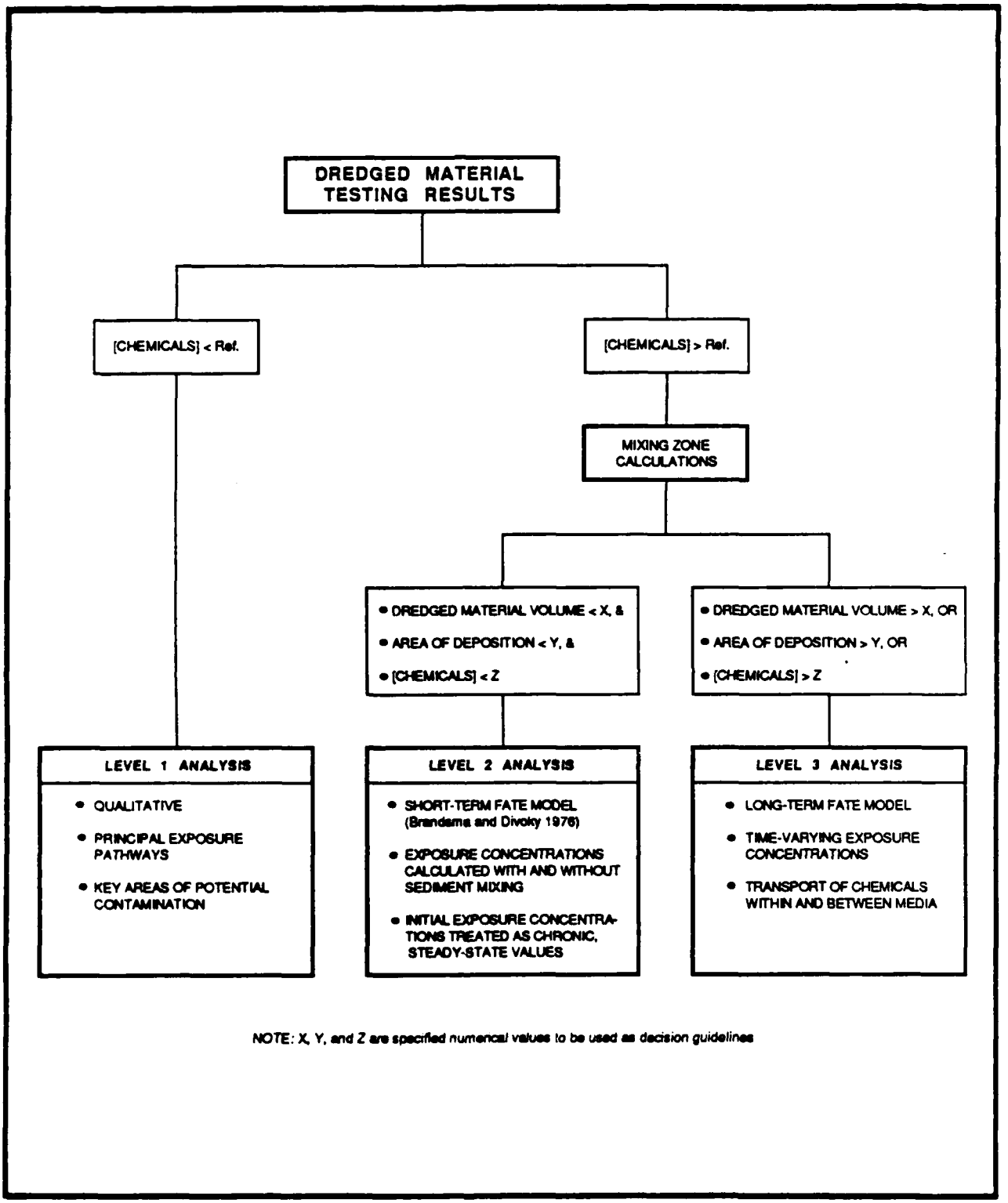


Figure 2. Extent of transport and fate analysis for chemical contaminants.

- Water quality criterion for the chemical, if available
- Volume of dredged material discharged.

U.S. COE (1985) provides methods for calculating mixing zones associated with aquatic disposal of dredged material, upland disposal of effluent from the dredged material, and surface runoff from an upland disposal site.

Level 1 Analysis--A qualitative analysis of transport and fate would be sufficient when the results of sediment testing indicate no significant elevation of test variables above reference conditions. Level 1 analysis considers information on physical-chemical properties of contaminants (see Hazard Identification above), relevant site characteristics, and potential exposure pathways. The product of this analysis is a determination of principal exposure pathways and potential points of environmental contamination for each chemical of concern.

Level 2 Analysis--For deepwater marine disposal options, the sediment transport and fate model (Brandsma and Divoky 1976; Johnson 3 March 1986, personal communication) discussed earlier is the basis for a Level 2 analysis to predict environmental concentrations of contaminants in bottom sediments. The model yields an estimate of the bottom area covered by dredged material. The average concentration of a contaminant in the dredged material sample tested is assumed as a constant over the entire area (Figures 3B and 4A).

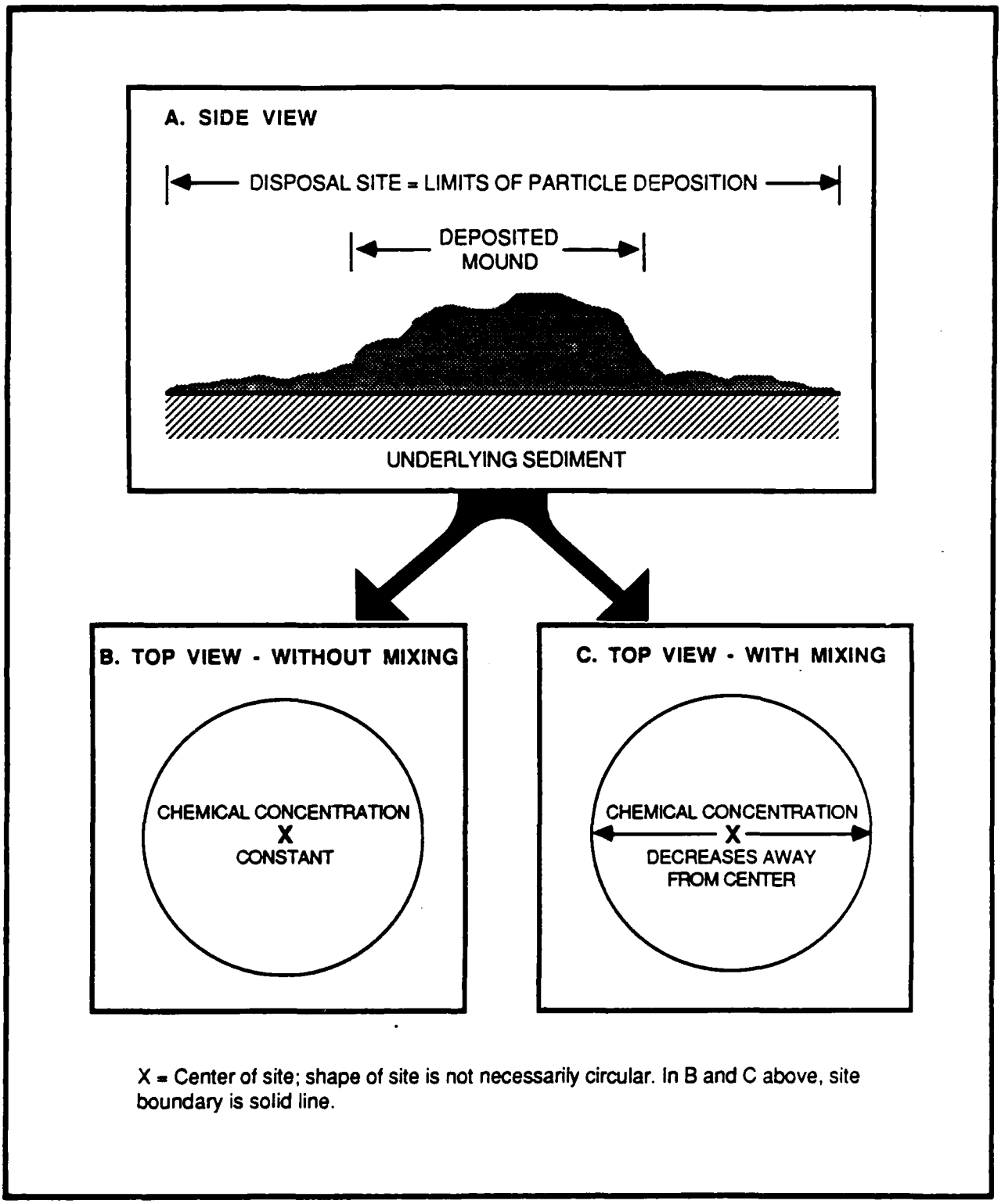
To obtain an upper estimate of exposure concentration, the following assumptions are adopted:

- Bulk sediment concentrations of contaminants obtained from laboratory analyses are used as estimates of final concentrations in deposited sediments (i.e., contaminant concentrations are assumed to be independent of particle-size composition and each contaminant is treated in a mass-conservative manner with no phase changes)
- Mixing of contaminated sediments with underlying sediments does not occur
- Newly deposited sediments remain in place and are not resuspended.

To obtain a lower-bound estimate of exposure concentrations, mixing is assumed to occur to the depth of bioturbation in Puget Sound (10 cm; Carpenter et al. 1985). Thus, average concentrations of contaminants in the mixed layer can be estimated as:

$$C_i(x) = \frac{D(x) \cdot CS_i}{d} \text{ for all } D(x) < d \qquad (1)$$

$$C_i(x) = CS_i \text{ for all } D(x) \geq d \qquad (2)$$



X = Center of site; shape of site is not necessarily circular. In B and C above, site boundary is solid line.

Figure 3. Hypothetical patterns of sediment deposition and resulting concentrations of a contaminant after dredged material disposal.

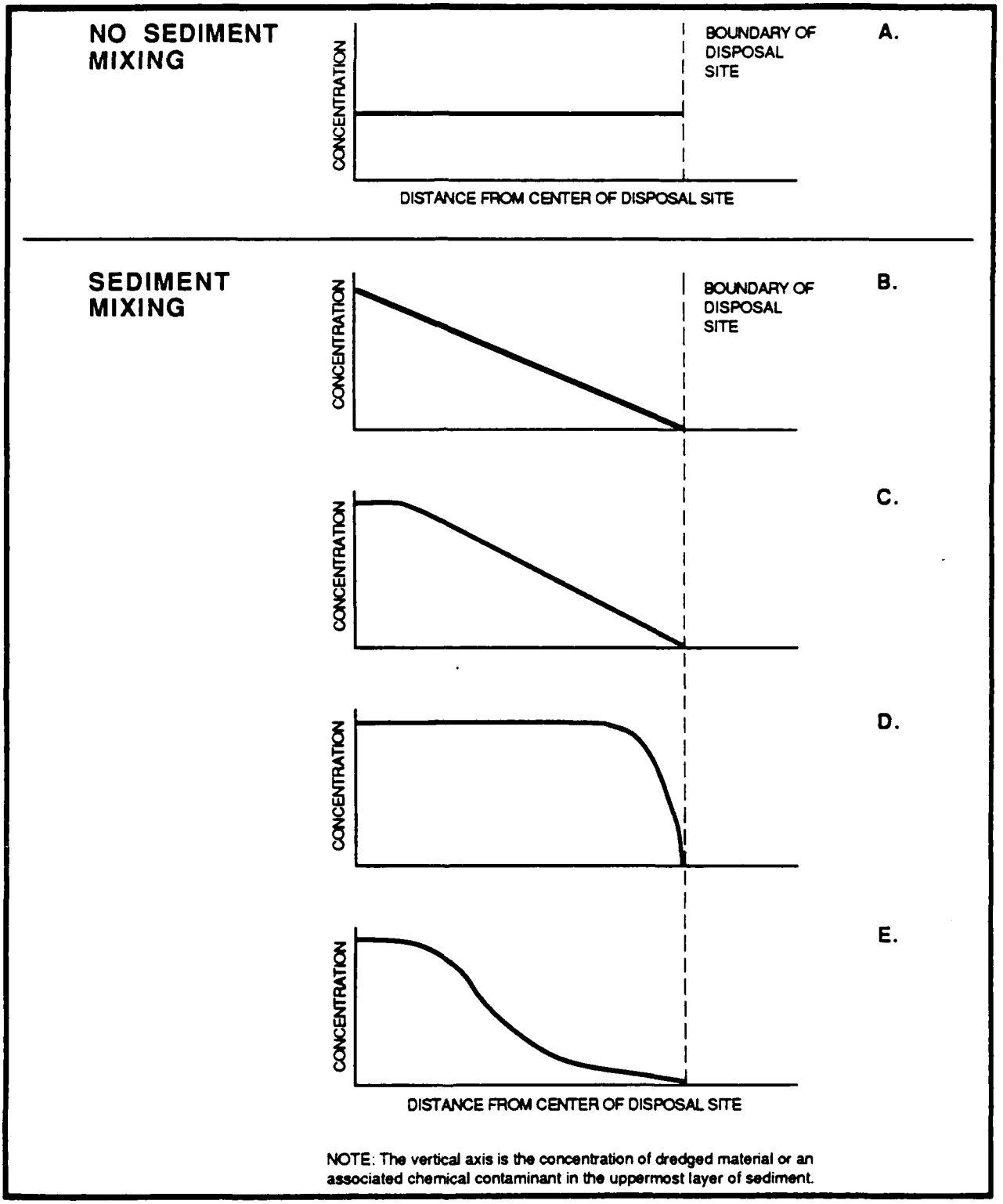


Figure 4. Possible spatial distributions of dredged material and associated contaminants at an unconfined deep-water site.

where:

$C_i(x)$ = Average concentration of chemical i in the mixed layer at distance x from the center of the disposal site

$D(x)$ = Depth of deposited sediment at distance x from the center of the disposal site

CS_i = Measured concentration of chemical i in dredged material

d = Depth of mixed layer.

This model assumes that pre-disposal concentrations of contaminants at the dispoal site are small relative to contaminant concentrations in dredged material.

Using a discrete form equation to integrate over distance, the average concentration for the entire disposal area is therefore:

$$\bar{C}_i = \frac{1}{\sum_x A(x)} \sum_x [C_i(x) \cdot A(x)] \qquad (3)$$

where:

$\bar{C}_i$ = Average concentration of chemical i in the mixed layer of the entire disposal site

$A(x)$ = Area with average concentration $C_i(x)$ at distance x from the center of the disposal site

The first area considered is bounded by a circle with its center at the center of the disposal site. Subsequent areas are concentric doughnut-shaped areas. The average concentration in the inner circular area divided by the average concentration in the outermost area may be used as an index of the spatial pattern of concentration, exposure, and risk. The depth of deposited sediment [$D(x)$] and therefore the predicted concentration [$C_i(x)$] decrease with distance from the center of the disposal site (Figures 3C and 4B-E). Both upper-bound and lower-bound estimates of exposure concentrations are treated as steady-state values for subsequent use in exposure assessment.

For each contaminant (or bulk dredged material), the exposure concentration decreases with distance from the center of the disposal site if sediment mixing is assumed in the model of transport and fate. An assumption of no mixing leads to 100 percent dredged material at the sediment surface, where organisms contact is assumed to take place. Some possible spatial distributions of sediment/contaminant concentrations are shown in Figure 4. For purposes of modeling, it may be possible to equate spatial patterns C and E with pattern B and pattern D with pattern A.

Concentration vs. distance curves are dependent on the degree of bioturbation of the deposited matter. For example, in panel A, the assumption of "no bioturbation" leads to a contamination pattern that reflects lack

of sediment mixing. Although the thickness of deposited sediment decreases with distance from the center of the disposal site, the concentration of a chemical constituent (or the percentage of dredged material) in the uppermost sediment remains constant. Note that the effect of surface area grain-size relationships may actually result in an increase in concentration (on a dry-weight basis) with distance. This effect is expected to be small (i.e., a factor of 2-3 at most). Moreover, in cases of sediment mixing, the thickness of deposited coarse sediments near the center of the site relative to that of fine sediments at the periphery of the site would overwhelm the effect of grain-size composition on contaminant concentrations.

For water-column exposures, the Level 2 analysis of transport and fate follows a similar approach to that just described for deposited sediment. Because of rapid dilution of dredged material plumes in water, the analysis is based on the mixing zone calculation described by U.S. COE (1985). As before, chemical concentrations are related to suspended sediment concentrations.

A Level 2 analysis of chemical fate at nearshore and upland sites might be restricted to quantitative modeling of transport through principal environmental pathways (see above, Exposure Scenarios). Thus, the simulation models are a subset of more complex models used for Level 3 analysis discussed in the next section. The results of Level 2 analysis might be steady-state concentrations for various media in the project area.

Level 3 Analysis--Level 3 analysis requires more complex modeling of environmental fate and transport. For example, all potential exposure pathways may be included in simulations. Chemical transfer between media and chemical transformations as well as transport within each medium would be included in the model. A wide variety of fate and transport models is available for use in exposure assessment (Mills et al. 1983; Burns et al. 1981; Games 1983; Onishi 1985a,b). A review of available models and recommendation of specific models is beyond the scope of this effort. The choice of model will depend on the level of complexity and detail needed for a particular project. One particular modeling system of note is U.S. EPA's Graphical Exposure Modeling System (GEMS) which can be accessed through the U.S. EPA regional offices. GEMS includes a series of models useful for assessing the fate of chemical contaminants in air, surface water (and sediments), groundwater, and soil (U.S. EPA 1982). The output from a detailed simulation model would be estimates of concentrations of contaminants of concern over space and time.

The analysis of transport and fate mechanisms concludes the first stage of the exposure assessment. The next stage is to characterize the exposed populations.

Exposed Population Analysis

The second stage of the exposure assessment, analysis of exposed populations, includes the following steps for each disposal option:

- Identify potentially exposed populations and map location relative to disposal site

- Characterize potentially exposed populations including identification of sensitive groups (e.g., infants, elderly, and pregnant individuals in human populations; endangered or sensitive wildlife species; commercial and recreationally harvested species), and estimation of population abundance and sex/age composition if the latter is relevant to interpretation of toxicological effects

- Analyze population activities such as movement patterns, time spent at or near disposal site, and dietary habits

- Estimate exposure coefficients in terms of average amount of contaminated medium contacted per day (e.g., daily intake of drinking water, daily intake of fish, inhalation rate).

Populations selected for study may initially be chosen on the basis of proximity to the contamination source, but additional populations at risk will be identified on the basis of transport and fate modeling, existing data on sensitive populations, and information on harvest of aquatic or terrestrial species.

A list of exposure coefficients that may be required for human exposure assessments is provided in Table 3. A media-specific exposure coefficient is combined with an estimate of contaminant concentration in the same medium to calculate exposure level, as discussed in the next section.

Integrated Exposure Analysis

In the integrated exposure analysis, information on estimated contaminant concentrations and exposed population characteristics are combined to develop an exposure profile for each exposure pathway and route. The exposure profile may include an estimate of exposure level and dose. The exposure level may be defined as the rate of contact with contaminant via a specific route (dermal, inhaled, oral). The dose is the rate of absorption of contaminant that equals the exposure level multiplied times a route-specific absorption coefficient. Total exposure dose is obtained by summing estimates of exposure doses for each route. As explained below, exposure analyses differ slightly depending on whether the population of concern is human or nonhuman.

The extent of integrated exposure analysis depends on the results of the model(s) of transport and fate. A Level 1 analysis of transport and fate would lead to a Level 1 analysis of exposure. A Level 2 analysis of transport and fate would yield estimates of average media concentrations for zones within the project area or for the area as a whole. Conservative estimates of exposure would result from treating the media concentrations predicted by a Level 2 analysis as steady-state values. Finally, a Level 3 analysis of transport and fate would give estimates of media concentrations over space and time. Estimates of exposure would then be treated as time-dependent variables integrated over space.

TABLE 3. STANDARD VALUES OF EXPOSURE COEFFICIENTS USED IN DAILY INTAKE CALCULATIONS

Coefficient	Standard Value	Reference
Average body weight, adult	70 kg	U.S. EPA 1980b
Average body weight, child	10 kg	ICRP (1975)
Water ingestion rate, adult	2 L/day	NAS (1977)
Water ingestion rate, child	1 L/day	NAS (1977)
Inhalation rate, adult	20 m^3/day	U.S. EPA (1980b)
Inhalation rate, child	5 m^3/day	U.S. FDA (1979)
Fish consumption rate, adult	6.5 g/day	U.S. EPA (1980b)

Reference: ICF (1985).

Level 1 Analysis--

The considerations of a Level 1 analysis of exposure would not depend on whether the potentially exposed population is human or nonhuman. In this qualitative analysis, the following factors are summarized:

- Distribution and fate of dredged materials and contaminants
- Principal pathways and routes of exposure
- Characteristics of potentially exposed populations
- Estimates of contaminant concentrations in media contacted by potentially exposed populations
- Estimates of exposure levels for principal pathways and routes.

Level 2 Analysis--

A Level 2 analysis is applied separately to humans and nonhuman species. The methods used for each procedure differ substantially.

Human Populations--An example format for the display of an integrated exposure analysis for a hypothetical human population is shown in Table 4. The table format allows storage of exposure information in a computer spreadsheet, facilitating calculation of route-specific doses and associated risks. Columns of notes containing references to sources of information can easily be added to the spreadsheet to further document the exposure analysis. The following factors may enter into a calculation of an exposure value (dose):

- Contaminant concentrations in the medium of concern (e.g., soil, water, air, fish muscle)
- Conversion factors (to account for dilution or other processes that affect the effective exposure concentration)
- Rate of contact with the contaminated medium (e.g., liters of water consumed, cubic meters of air inhaled, grams of soil ingested)
- Exposure duration (expressed in absolute units, such as days or years, or as relative units, such as fraction of 70-yr lifetime during which exposure may occur)
- Absorption coefficient [expressed as the relative ratio of contaminant assimilation indices (contaminant mass absorbed divided by contaminant mass ingested for humans relative to that for laboratory bioassay animals)]
- Body weight of target organisms (to account for dilution of the absolute mass absorbed within the body mass of the target organism).

TABLE 4. EXAMPLE FORMAT FOR DISPLAY OF INTEGRATED EXPOSURE ANALYSIS FOR HUMAN POPULATIONS

SCENARIO
(description of major assumptions)

	Exposure Determination									Risk Determination		
Route of Exposure	Concentration in Medium (mg/kg)	Adjusted Concentration (mg/kg)	Contact Rate (g/day)	TOTAL DAILY CONTACT (mg/day)	Exposure Duration (years)	Exposure Duration Modifier (0-1.0)	Absorption Coefficient (0-1.0)	Body Weight (kg)	EXPOSURE VALUE (mg/kg/d)	Potency Factor 1/(mg/kg/d)	RISK	Percent Total Risk
ORAL ROUTES:												
Food Ingestion												
(food source 1)					70.00	1.00	1.00	70.00				
(food source 2)					70.00	1.00	1.00	70.00				
(food source 3)					70.00	1.00	1.00	70.00				
(food source 4)					70.00	1.00	1.00	70.00				
(food source 5)					70.00	1.00	1.00	70.00				
Drinking water					70.00	1.00	1.00	70.00				
Soil Ingestion												
0-9 mo					0.75	1.00	1.00	4.46				
9-18 mo					0.75	1.00	1.00	7.10				
1.5-3.5 yr					2.00	1.00	1.00	11.94				
3.5-5 yr					1.50	1.00	1.00	18.10				
5+ yr (5-18 yr?)					13.00	1.00	1.00	43.62				
(18-70 yr?)					52.00	1.00	1.00	70.00				
INHALATION ROUTE:												
Dust Inhalation												
inhaled fraction					70.00	1.00	0.75	70.00				
ingested fraction					70.00	1.00	0.50	70.00				
Gas/Vapor Inhalation					70.00	1.00	1.00	70.00				
DERMAL ABSORPTION ROUTE:												
Dermal Absorption					70.00	1.00	1.00	70.00				
											Sum	

It should be emphasized that some of these variables are capable of being measured with great precision (e.g., contaminant concentrations in water), whereas others may only be estimated on an order-of-magnitude basis. The precision and accuracy of the final risk estimates are directly related to the precision and accuracy of the variables incorporated into the model equations. Uncertainty is easily tracked with a spreadsheet format by calculating exposure estimates for low, mid, and high values of key variables within their respective plausible ranges. Specification of probability distributions for key variables is an alternative method of uncertainty analysis (see below, Uncertainty Analysis).

The predicted human exposures derived for risk analysis of dredged materials disposal options are necessarily estimates of hypothetical exposures. Most analyses of human exposure for evaluation of disposal options will have the following characteristics (corresponding to a Level 2 analysis of transport/fate):

- Standard estimates of variables, such as fish consumption rates, will be used because of lack of site-specific information on the exposed population
- The size and locations of the exposed population will be unknown or based on predictions from models of transport and fate of sediments/contaminants
- Concentrations of contaminants in various media will be based on direct extrapolation of residues obtained from laboratory tests [e.g., sediment, elutriate, and tissue chemistry (U.S. COE 1985)], assuming steady-state conditions. If transport and fate model results include estimates of media concentrations at various distances from the center of a disposal site, sets of exposure calculations (e.g., Table 4) would be generated for various locations
- Probability of exposure will be assumed to be 1.0 for each area of dredged material disposal/deposition where human contact is expected. However, if stochastic simulations of transport and fate analysis are conducted, it would be possible to estimate human exposure probabilities as the probability that some portion of the disposal site will overlap with a specified human-use area
- Only principal exposure pathways and routes will be addressed
- Consequently, exposure estimates will be hypothetical values, not estimates of actual exposures.

Because the analysis would be conducted before the disposal operation, exposure must be predicted. Monitoring of exposures after disposal may be desirable, and the information gained would certainly be useful in analyses of similar projects in the future. It is assumed that such monitoring information will not be available in the vast majority of cases. Furthermore,

information on characteristics of the exposed population (e.g., eating habits, activity patterns) is likely to be unavailable.

Nonhuman Populations--A Level 2 integrated exposure analysis for nonhuman populations provides quantitative estimates of exposure concentrations, assuming steady-state conditions based on short-term fate of dredged material and associated contaminants. The analysis summarizes the results from Level 2 modeling of transport and fate, which may include spatial patterns of contamination. The exposure analysis considers migrant and nonmigrant species separately. The methods are essentially the same for all disposal options (i.e., upland vs. nearshore vs. unconfined, deepwater disposal).

Various measures of exposure are possible, depending on the characteristics of the biological resource, the available data or distribution of the resource, and the results of the transport/fate model. The scenarios discussed below are based on deepwater unconfined disposal, but analogous measures of exposure generally apply to similar exposure pathways for upland and nearshore disposal sites. Note that biological effects for nonhuman populations are often related to exposure concentrations rather than exposure doses (as in humans). Therefore, exposure estimates for nonhuman populations are based on the probability of exposure to specified concentrations of dredged material or contaminants at specified locations.

The approach to estimating the probability of exposure depends on the kind of biological resource (see below). Examples of general categories of biological resources relative to deepwater disposal include:

- **Nonmigrant** organisms with a **ubiquitous** population distribution in space (e.g., infaunal amphipods). These organisms do not move relative to the disposal site. They may occupy diverse habitats within and outside of the site. On a fine-scale, these organisms may be restricted to specific kinds of sediment.

- **Nonmigrant** organisms with a **discrete** population distribution in space (e.g., infaunal or epifaunal shellfish such as oysters and geoducks). These organisms do not move relative to the disposal site. They often occur in dense aggregations, which exhibit a patchy distribution.

- **Large-scale migrant** organisms, with a ubiquitous population distribution in space (e.g., pelagic fish such as Pacific cod). These organisms move over relatively large areas relative to the size of the disposal site. It is assumed that the "home range" of an individual fish, which includes the entire disposal site, may be approximated by a value from the literature, or the area of the bay in which the disposal site is located. [Note that home ranges for terrestrial wildlife are well known relative to those for fish.]

- **Small-scale migrant** organisms with a ubiquitous population distribution in space. These organisms move over large areas relative to the disposal site, but their distribution may only partially overlap the disposal site.

The difference between a migrant and nonmigrant species is that an individual of the former moves in and out of the disposal area, whereas the nonmigrant individual does not. Movements of migrant individuals within their home range are assumed to be random.

The general model of average exposure probability for individuals of a **migrant species** in contact with bottom sediments is:

$$PE_{xr} = \frac{O_{xr}}{A_r} \tag{4}$$

where:

PE_{xr} = Probability of exposure of resource r to sediments at location x

O_{xr} = Area of overlap between distribution of resource r and location x

A_r = Total area occupied by exposed portion of resource r (total area of contiguous, individual home ranges that overlap any area of disposal site).

For modeling purposes, the home range of a migrant species may be stratified by habitat type. For example, the range of movement of an individual English sole includes many different habitats, especially during seasonal migrations. However, most of an individual's time is spent in habitats with fine-grained, muddy sediments. Therefore, the area of the home range may be considered as the total area of fine-grained, muddy sediments within the larger area of movement. The average total probability of exposure of a migrant individual to all disposal site areas is:

$$PE_r = \sum_x PE_{xr} = \frac{1}{A_r} \sum_x O_{xr} \tag{5}$$

where:

PE_r = Total probability of exposure of resource r for all areas within the disposal site.

Analogous exposure models may be derived for water-column exposures by substituting terms for volumes of water-column habitat for the corresponding habitat-area terms in the equations above.

For a **nonmigrant species**, the individual probability of exposure (PE_{xr}) equals 1.0 for all areas within the disposal site and 0.0 for all areas outside the disposal site. Average exposure probability ($\overline{PE}_{xr}$) is calculated in place of total exposure probability (PE_r) for a nonmigrant species.

Level 3 Analysis--

The Level 3 exposure analysis extends the previous models by considering changes in contaminant concentrations over time due to long-term transport, and transformation and degradation processes. If the predicted frequency

of disposal events is substantially less than the period over which these long-term processes operate, it may be desirable to incorporate multiple disposal events into the transport and fate model. The results of such models are estimates of concentrations of dredged material or contaminant in various media (see above, Estimation of Environmental Concentrations). These results may be summarized along with exposed population characteristics and exposure probabilities. In a Level 3 analysis, the models for estimating the probability of exposure for various biological resources may be the same as those discussed earlier under Level 2 analysis. More complex models of animal distribution and movement are available in the ecological literature, but their application is limited by the availability of data on actual movements of specific species.

RISK CHARACTERIZATION

In the risk characterization stage, results of the exposure assessment and the dose-response assessment are combined to estimate the probability and extent of adverse effects associated with contaminant releases. An overview of the risk characterization process is shown in Figure 5. Major assumptions, scientific judgments, and uncertainties are also summarized. If the exposure assessment includes only qualitative estimates of environmental concentrations of dredged material and contaminants (Level 1 analysis), estimates of risk will necessarily be qualitative. The general models for calculation of quantitative estimates of risk from exposure and dose-response data are similar for Level 2 and Level 3 analysis. In Level 2 analysis, risks are estimated from initial exposures that are assumed to be constant over the long-term (e.g., 70-yr human lifetime). In Level 3 analysis, estimated risks are based on integration of time-variable exposures. Quantitative models are addressed in the following sections.

Human Health Risk

In human health risk assessment, carcinogens and noncarcinogens are treated separately. Indices of risk for these different categories of toxicants are based on different dose-response models (see above, Dose-Response Assessment). Tetra Tech (1986b) provides detailed guidance for assessment of human health risk associated with consumption of chemically contaminated seafood.

Carcinogenic Risk--

Numerical estimates of carcinogenic risk can be presented in one or more of the following ways (U.S. EPA 1986a):

- Unit risk - the risk corresponding to a unit exposure of mg contaminant/kg body weight/day
- Dose or concentration corresponding to a specified level of risk - for example, a guideline for maximum allowable contamination of a specified medium may be derived by assuming a maximum allowable risk

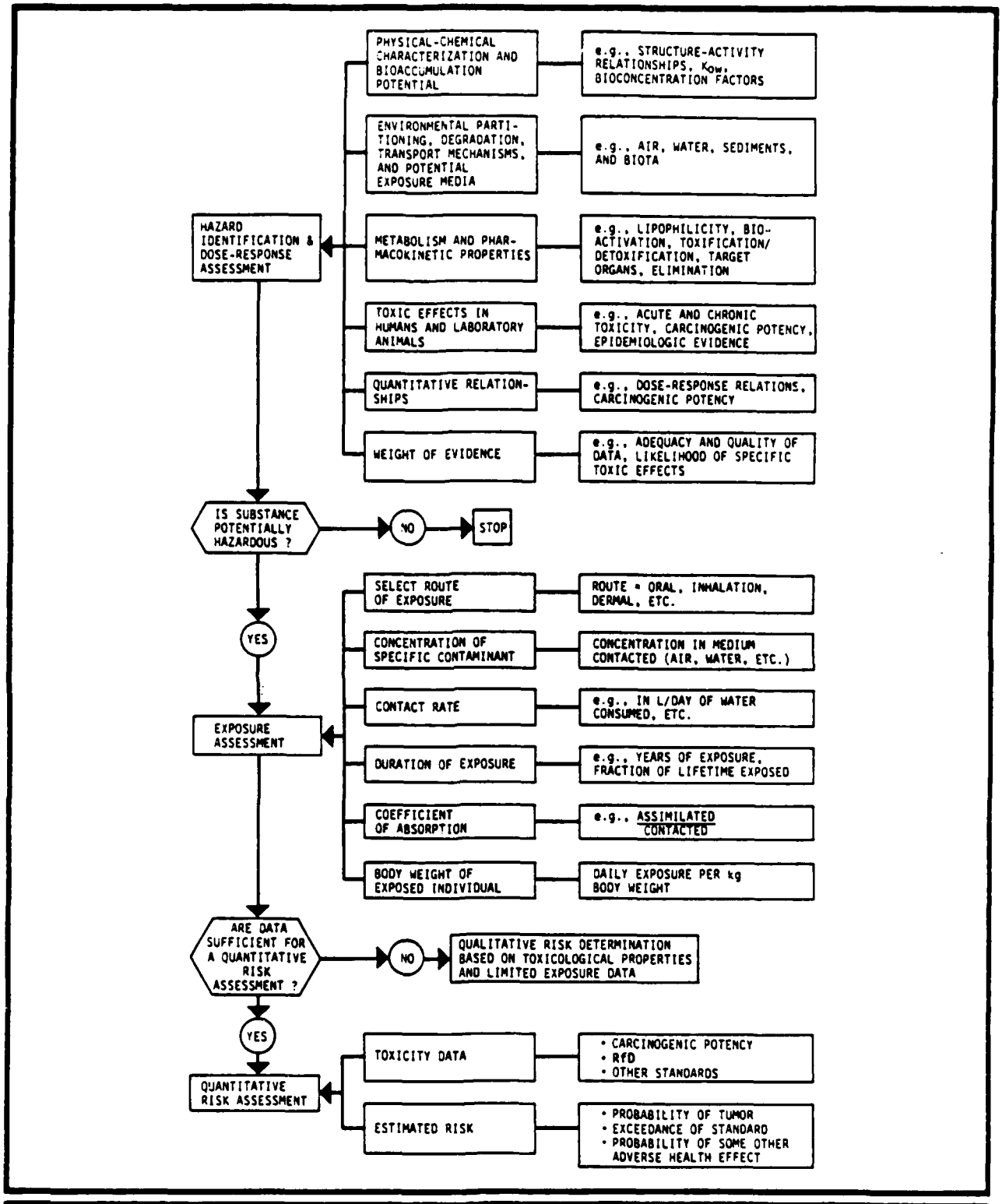


Figure 5. Conceptual structure of quantitative health risk assessment model.

- Individual and population risks - estimates of excess lifetime cancer risk may be expressed for individuals (as a probability estimate) or for the exposed population (as an estimate of the number of cancers produced per time).

Unit risks are useful for ranking chemical contaminants according to their carcinogenic potencies. As shown later, they are also involved in calculations of other numerical estimates of risk. Both contaminant concentration guidelines and individual lifetime risks are useful for dredged material risk assessment. Because extensive data on the exposed population are not likely to be available for dredged disposal evaluations, estimating population risks will usually be impossible. Regardless of the option chosen for expressing risk, final numerical estimates should be presented as one significant digit only (U.S. EPA 1986a).

General Model--The general model for estimating excess lifetime carcinogenic risk is:

$$PR_{ijk} = B_{ij} \cdot E_{ijk} \quad (6)$$

where:

$$E_{ijk} = \frac{C_{ik}\ I_{jk}\ X_{ijk}}{W} \quad (7)$$

and:

PR_{ijk} = Excess lifetime risk of cancer from intake of contaminant i via route j and medium k

B_{ij} = Unit risk or carcinogenic potency factor for contaminant i and exposure route j $[(mg \cdot kg^{-1} \cdot day^{-1})^{-1}]$

E_{ijk} = Average lifetime exposure dose of chemical i via route j and medium k $(mg \cdot kg^{-1} \cdot day^{-1})$

C_{ij} = Average concentration of chemical i in medium k (mg/kg)

I_{jk} = Contact rate (e.g., ingestion or inhalation rate) with medium k via route j (kg/day)

X_{ijk} = Absorption coefficient for uptake of chemical i via route j and medium k (dimensionless)

W = Reference human weight = 70 kg.

Combinations of media and routes were summarized earlier (see above, Exposure Assessment, Potential Exposure Pathways and Routes). The sources of estimates for these variables are summarized in Table 5. The results of the exposure and risk calculations would be displayed in the format shown in Table 4 above.

TABLE 5. SOURCES OF ESTIMATES FOR VARIABLES USED TO CALCULATE CARCINOGENIC RISK

Variable[a]	Definition	Source of Estimate
B_{ij}	Carcinogenic potency factors	U.S. EPA (1980b; 1985a)
C_{ij}	Contaminant concentrations	Laboratory measurement made on dredged material, or estimated environmental concentration based on transport/fate model
I_{jk}	Media contact rates, e.g., ingestion or inhalation rate	Standard coefficients (see Table 3 above) or estimate based on survey of exposed population
X_{ijK}	Absorption coefficients	Assumed equal to 1.0 unless data are available[b]
W	Human body weight	Reference body weight = 70 kg adult male (U.S. EPA 1980b). Means and percent distributions of body weights for males and females of various ages are given by Anderson et al. (1985).

[a] Subscripts are explained in text.

[b] This assumption requires that the assimilation index (amount of contaminant absorbed by an organism divided by amount ingested) for humans is equal to that of the laboratory bioassay animal used to determine carcinogenic potency factors. This is not equivalent to assuming that all contacted contaminant is absorbed by the human system.

Indices for Evaluation of Disposal Options--The human health risk estimates to be used in comparative analysis of disposal options may be based on excess lifetime risks associated with individual chemicals of interest, or on an estimate of total cancer risks. If individual chemical risks are used, the PR_{ijk} risk estimates are summed over all routes (j) and all media (k). Total risk of cancer at any target organic site may be approximated as the sum of all PR_{ijk} across all chemicals (i), all routes (j), and all media (k). These additive-model estimates should be viewed only as a rough indices of total risk (U.S. EPA 1986d). Regardless of the indicators used, the final risk estimate should be regarded as a plausible upper limit on the actual risk under the assumed conditions (U.S. EPA 1980b, 1986a). Despite realistic estimates of C_{ij} and I_{jk}, the potency factors (B_{ij}) derived by U.S. EPA are upper bounds on the upper 95 percent confidence interval of the slope of the dose-response curve for the most sensitive animal species/organ. Also, each absorption coefficient (X_{ijk}) is assumed to be equal to 1.0 unless data on actual values are available.

Noncarcinogenic Effects--

Current methods for predicting human health effects from exposure to noncarcinogenic chemicals rely on the concept of an Reference Dose (RfD), an average daily exposure that is considered safe or acceptable (Vettorazi 1976, 1980; U.S. EPA 1980b). The RfD is derived from an observed threshold dose (e.g., No Observed Adverse Effect Level) in a chronic animal bioassay by applying an uncertainty factor, which usually ranges from 1 to 1,000 (Douson and Stara 1983). The RfD is essentially the maximum lifetime exposure that is likely to lead to a negligible risk of the health effect of concern.

General Model--Because the slope of the dose response curve is not presently considered in deriving RfDs (Dourson et al. 1985), a formal, precise measure of noncarcinogenic risk is not available. Nevertheless, an index of noncarcinogenic risk may be approximated as the ratio of the estimated exposure to the RfD as follows:

$$HI_{ij} = \frac{1}{RfD_{ij}} \left(\sum_{k} E_{ijk} \right) \qquad (8)$$

where:

HI_{ij} = Hazard Index, or relative probability of a health effect from intake of chemical i via route j (dimensionless)

E_{ijk} = Estimated average lifetime exposure dose of chemical i from medium k via route j ($mg \cdot kg^{-1} \cdot day^{-1}$) calculated as defined earlier

RfD_{ij} = Reference Dose of chemical i via route j ($mg \cdot kg^{-1} \cdot day^{-1}$)

Where the index HI_{ij} is less than 1.0, no hazard is assumed (Stara et al. 1983; U.S. EPA 1986d). The magnitude of HI_{ij} for all values above unity defines the relative magnitude of the hazard. The RfD values for selected chemicals are available from U.S. EPA (1980b, 1986e; see Appendix C). Most RfDs are based on oral exposure. When the mode of toxicologic action is the same regardless of exposure route, it may be possible to extrapolate the

RfD from one route to another. However, the experimental basis for the RfD should be examined to ensure that absorbed dose was measured. Otherwise, one may have to correct for differences in absorption efficiency among routes.

Indices for Evaluation of Disposal Options--As was discussed earlier for carcinogens, the noncarcinogenic health risk indicator(s) to be used in evaluating dredged material disposal options may be based on indices (HI_{ij}) for selected individual chemicals or a summation of indices for multiple chemicals. For chemicals without route-specific effects, the absorbed doses are additive among routes (U.S. EPA 1986d). For example:

$$E_i = \sum_j \sum_k E_{ijk} \tag{9}$$

and:

$$HI_i = \sum_j HI_{ij} \tag{10}$$

The sum of hazard indices for multiple noncarcinogenic chemicals is:

$$HI_T = \sum_i HI_i \tag{11}$$

where:

HI_T = Hazard index for the chemical mixture.

The hazard index HI_T is compared to unity to evaluate qualitatively the hazard resulting from exposure to the chemical mixture (U.S. EPA 1986d). U.S. EPA (1986d) and the American Conference of Governmental Industrial Hygienists (ACGIH 1981) recommend this approach. Although ACGIH (1981) recommends against addition of hazard indices for individual chemicals known to act on different large organs, U.S. EPA (1986d) notes that it may be appropriate to presume that hazards are additive when information on chemical interactions is lacking.

Derivation of Contaminant Concentration Guidelines--

The risk assessment models just described have been used to set contaminant concentration guidelines; e.g., regulatory criteria or waste site cleanup goals. In the guidelines-setting approach, a target (or acceptable) risk level is defined by policy and the corresponding allowable concentration guideline is calculated for a chemical or group of chemicals. U.S. EPA (1980b) used this approach to develop water quality criteria although the reference risk levels were not interpreted as safe or acceptable levels of carcinogens. For a noncarcinogen, the RfD is used in place of the reference risk level. Dacre et al. (1980) and Rosenblatt et al. (1983) used a similar approach to establish "preliminary pollutant limit values," [i.e., allowable concentrations of chemical contaminants in various media (air, water, soil)].

An example of the guidelines-setting approach is provided in Appendix A. In the example, guidelines are derived for maximum advisable concentrations

of contaminants in seafood. These tissue-contamination guidelines are useful for rapid interpretation of data from laboratory bioaccumulation experiments conducted as part of the U.S. COE testing strategy for dredged material. They are intended to be used in place of or in addition to action levels set by the U.S. Food and Drug Administration.

Preliminary pollutant limit values (Rosenblatt et al. 1983) or tissue contamination guidelines (Appendix A) are valuable for interpretation of contaminant data sets or for setting environmental quality criteria. Nevertheless, their use in comparative risk analysis of disposal options would result in an indirect assessment of risk. In the comparative analyses described later, direct estimates of potential human health risks are used based on the models presented above.

Ecological Risk

Approaches to ecological risk characterization described in this section differ from those of human health risk assessment in that:

- The probability of exposure is not necessarily assumed equal to unity for a location of concern
- Effects are measured in terms of organism mortality (or population extinction) rather than potentially sublethal responses such as cancer or birth defects
- Measureable responses to contaminant exposure are more likely to occur over a short-term period (days, weeks) due to direct exposure of nonhuman species to deposited dredged material
- Ecological risk is related to exposure concentration rather than dose
- The exposed nonhuman population is more easily characterized than is a human population.

The two levels of quantitative risk analysis (corresponding to Level 2 and Level 3 exposure analysis) are presented below.

Level 2 Analysis--

The model used to estimate area-specific risk to an individual based on Level 2 exposure analysis is:

$$PM_{xr} = PE_{xr} \cdot PB_{xr}\,[C(x)] \qquad (12)$$

where:

PM_{xr} = Probability (risk) of mortality of an individual of resource species r due to exposure at location (area) x.

PE_{xr} = Probability of exposure of an individual of species r at location x, as a function of resource overlap with location x (O_{xr}) and area occupied by resource r (A_r).

$PB_{xr}[C(x)]$ = Probability of mortality of an individual of species r upon exposure to concentration, C(x), of dredged material at location x. The relationship between probability of mortality and concentration is based on a dose-response curve from the sediment bioassay.

C(x) = Concentration (or percentage) of dredged material at location x.

A population impact index for a specific area is calculated as:

$$I_{xr} = PM_{xr} \cdot N_{xr} \cdot A(x) \tag{13}$$

where:

I_{xr} = Number of individuals killed due to exposure to dredged material at location x

N_{xr} = Average population density (no./area) of species r at location x

A(x) = Area of location x.

As before, the term x can be interpreted as average distance from the center of the disposal site where each location is considered a donut-shaped area. Note that only chemical impacts are addressed by Equations 12 and 13. To account for burial of sensitive species, the probability of mortality (PM_{xr}) of sensitive organisms would be set equal to 1.0 whenever the thickness of the sediment deposit exceeded a specified threshold.

For each individual of a **migrant species** r, the total individual risk is the cumulative probability of mortality, PM_r, which is calculated as follows:

$$PM_r = \sum_x PM_{xr} \tag{14}$$

and cumulative population impact I_r is:

$$I_r = PM_r \cdot N_r \cdot A_r \tag{15}$$

where:

I_r = Total number of individuals of species r killed by exposure to dredged material.

N_r = Average population density (no./area) within the total area occupied by species r. The product of the last two terms in Equation 15 yields the total number of individuals of species r in the exposed population.

A_r = Area occupied by resource species r.

For each individual of a **nonmigrant species**, the area-specific risk of individual mortality is by definition the total risk. Instead of cumulative

probability, the population impact index (I_r) is calculated based on the average individual risk, as follows:

$$\overline{PM}_{xr} = \frac{1}{\sum_x [N_{xr} \cdot A(x)]} \sum_x [PM_{xr} \cdot N_{xr} \cdot A(x)] \quad (16)$$

and:

$$I_r = \sum_x [PM_{xr} \cdot N_{xr} \cdot A(x)] \quad (17)$$

where:

$\overline{PM}_{xr}$ = Average probability of mortality for individual nonmigrant species r due to exposure to dredged material

and other terms are defined above.

The function $PB_{xr}[C(x)]$ is described by a linear regression fit to dose-response data. For example, for a sediment toxicity bioassay (e.g., with the amphipod Rhepoxynius abronius), the function $PB_{xr}[C(x)]$ may be derived by regression of percent mortality against percent of dredged material. Such data may be easily obtained in a series of bioassays using dredged material diluted with various proportions of clean sediment. Because laboratory bioassay results cannot be extrapolated directly to field conditions, all population impacts (I_r) calculated as above should be viewed as relative indices of population risk, not as precise estimates of the number of organisms killed.

For some resource species, dose-response data will not be available. In such cases, it may be appropriate to use an approximate index of relative risk such as exposure probability multiplied times population abundance, or exposure probability multiplied times exposure concentration. Although these approaches have obvious limitations, they may be useful as interim measures, especially for valuable resources, until more data can be collected.

Level 3 Analysis--

Temporal variation in contaminant concentrations in sediments at a disposal site requires a risk analysis framework capable of integrating time-variable exposures. Intermittent water-column exposures may be generated by environmental disturbances such as storm-induced resuspension of sediments at aquatic disposal sites. Experimental work on time-dependent processes such as contaminant adsorption/desorption, biological uptake and depuration, and contaminant resuspension by physical or biological processes is being conducted as part of the U.S. COE/U.S. EPA Field Verification Program (Gentile et al. 1983a; Bierman et al. 1986).

The purpose of this section is to briefly describe the framework for risk analysis of dredged material disposal options when spatial and temporal variation of contaminant concentrations are considered. Several approaches of varying complexity are explored below. The choice of an approach will

depend on site-specific conditions and the objective of the risk analysis (e.g., characterization of number of organisms killed per time vs. impact on dynamics of a population or community). Details of previous approaches to ecological risk analysis are provided by Parkhurst et al. (1981), Gentile et al. (1983a,b), Vinegar (1983), Gentile and Shimmel (1984), Bergman et al. (1986), and Bierman et al. (1986).

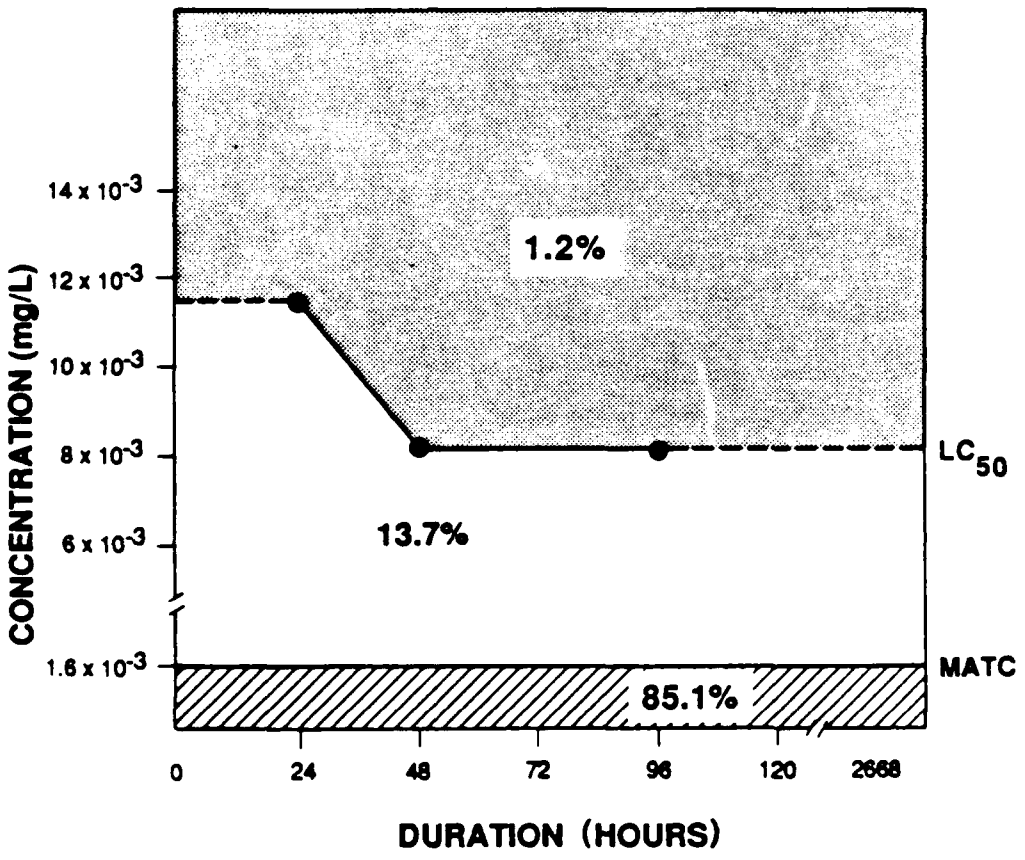
Four general approaches to a Level 3 risk characterization for a specified resource and a specified medium are outlined below. A common element of the various approaches should be the graphic display of variations in exposure concentrations of contaminants over time and space relative to resource distributions.

Approach 1: Time-Averaged Exposure and Risk--

- Use site-specific information on exposure to calculate exposure concentrations for each time interval (i.e., transport/fate model)
- Determine average exposure probability and concentration for each location (area) by integrating over time
- Use average exposure estimates and dose-response data (bioassay of given duration) to calculate individual and population risk (probability of mortality) according to Equations 12 and 13 above for each location
- Determine total risk by integrating over space
- Display plots showing variation of individual and population risk over time by species and location (or over all areas).

Approach 2: Frequency of Unacceptable Exposure--

- Use transport and fate model to estimate exposure concentrations for each time interval
- Determine median lethal concentrations (LC_{50}s) and Maximum Acceptable Toxicant Concentration (MATC) vs. duration of exposure based on bioassays conducted at various concentrations and exposure durations (Figure 6). The LC_{50} curve represents a short-term (acute) lethality reference variable. The MATC defines the sublethal no-effect boundary. The MATC usually corresponds to the lowest concentration capable of producing an effect in a long-term (chronic) bioassay. Methods for derivation of these curves are summarized in Parkhurst et al. (1981).
- Calculate risk as the percentage of time that the exposure concentration exceeds the MATC or the LC_{50}, based on a computer simulation. For example, as summarized in Figure 6 the concentration of a chemical of concern exceeded the MATC 13.7 percent of the time. The LC_{50} was exceeded 1.2 percent of the time. This approach can also be applied to bulk



Reference: Adapted from Parkhurst et al. (1981)

Figure 6. Summary of ecological risk analysis based on frequency of unacceptable exposures.

sediment rather than individual contaminants. Note that this method does not take into account the probability of exposure. However, it may be possible to incorporate exposure probability into the model by combining this method with procedures for calculating exposure probability discussed previously with respect to Level 2 analysis.

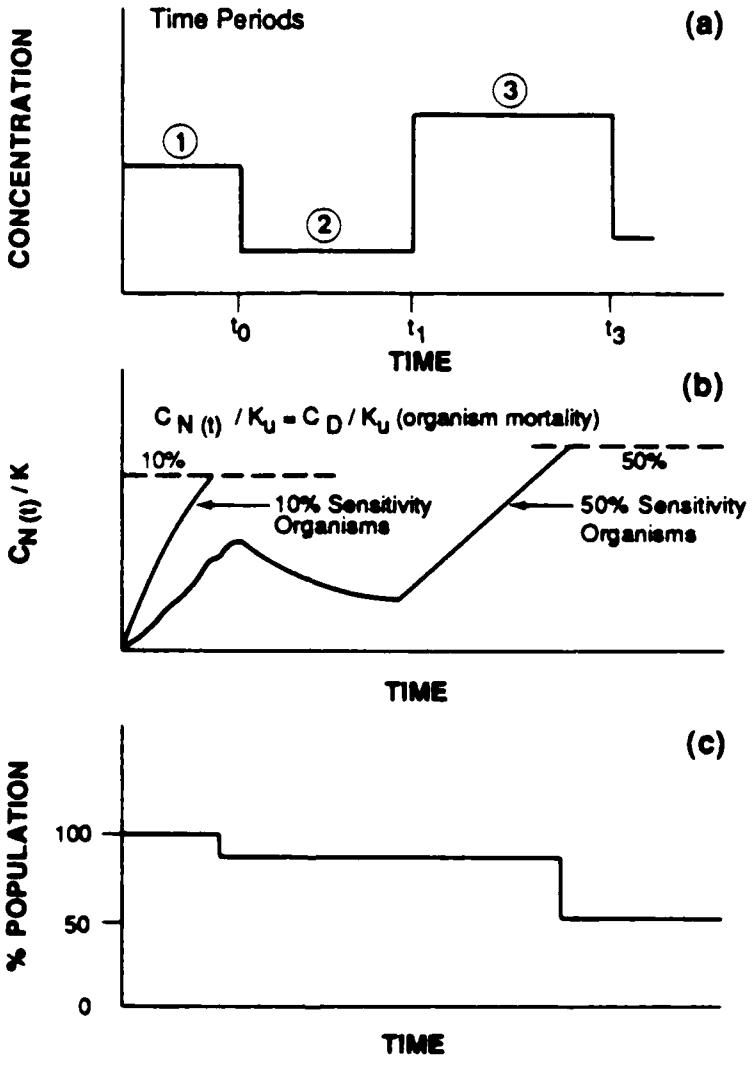
Approach 3: Time Variable Uptake and Depuration Kinetics--

- Use transport and fate model to estimate exposure concentrations of individual contaminants for each time interval
- Specify biokinetic model of contaminant uptake and depuration for species of concern (e.g., Mancini 1983)
- Determine the relationship between percent mortality and time for various exposure concentrations of contaminants
- Determine the cumulative risk due to exposure to time-variable concentrations of sediment based on the model of Mancini (1983). The cumulative population risk is equal to the total percentage of the initial population killed over a specified time period.

An example of this approach is shown in Figure 7. The upper panel shows variations in exposure concentration over time. The lower panel shows the percentage of the population remaining at each time interval t. The middle panel shows the temporal variation in the equivalent dose, defined as the body burden of contaminant [$C_{N(t)}$], divided by the toxicant uptake rate (K_u). When the equivalent dose after exposure period t is equal to the body burden at depth [C_D], mortality occurs. The model accounts for variation in sensitivity to the toxicant among individuals. The first 10 percent of the population that dies is termed the 10 percent sensitivity group. As shown in the middle panel of Figure 7, their body burden increases at a faster rate than the 50 percent sensitivity organisms. Also, the toxicant body burden associated with death may be lower for the more sensitive organisms.

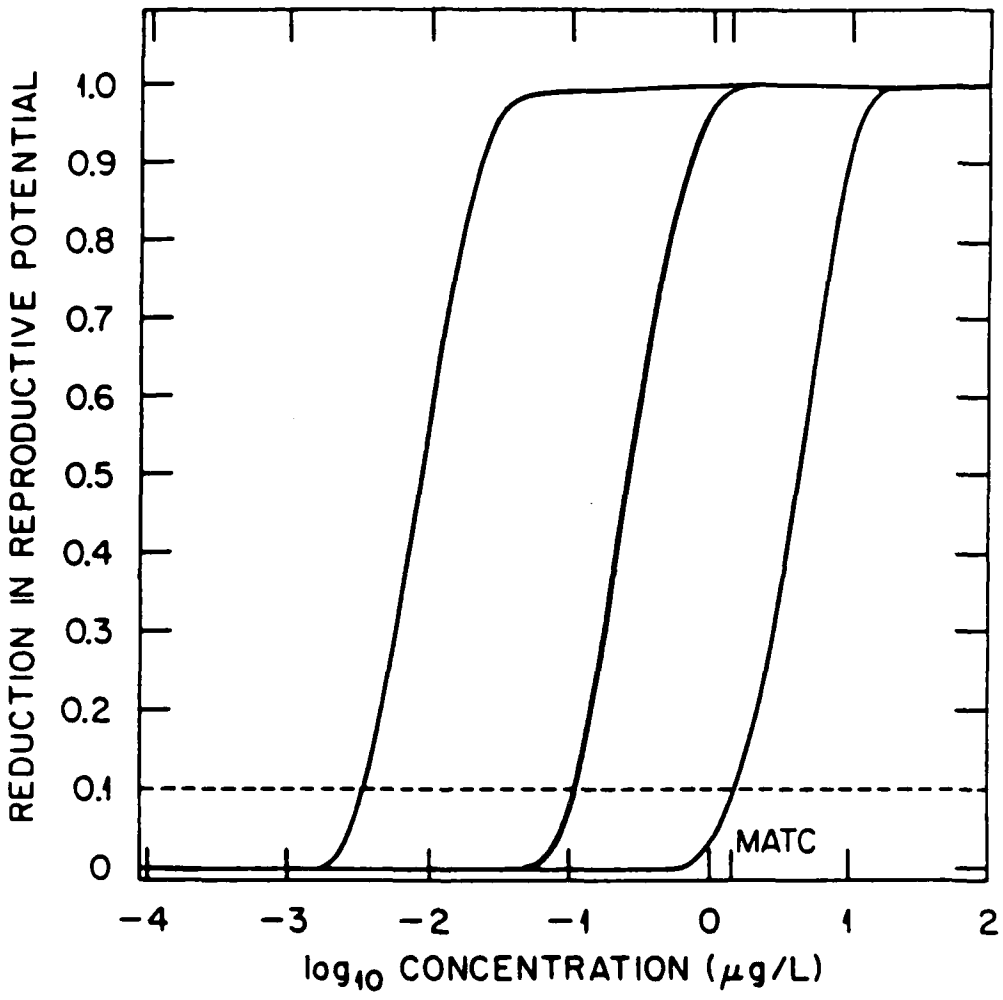
Approach 4: Population Modeling--

- Use transport and fate model to estimate exposure concentrations of individual contaminants for each time interval.
- From bioassay experiments, estimate effects of contaminants on age-specific fecundity and mortality rates, age at first reproduction, and other population variables.
- Using a life-table analysis (Daniels and Allan 1981; Gentile et al. 1982, 1983b) or a Leslie Matrix population model (Vinegar 1983), examine the effects of exposure to various concentrations of contaminants on the intrinsic rate of population increase (r) and age-specific reproductive values (V_a). An example of this approach provided by Dr. Lawrence Barnthouse at the workshop is shown in Figure 8.



Reference: Mancini (1983)

Figure 7. Example application of uptake and depuration kinetics model.



NOTE: Two outer curves are 95 percent confidence bands.
MATC (– – – – –) Maximum Acceptable Toxicant Concentration

Reference: Barnthouse (16 December 1985, workshop presentation)

Figure 8. Example of ecological risk analysis based on reduction in reproductive potential of a population.

- Using a dynamic simulation model linking the population dynamics model with the transport and fate model, estimate the percentage of time that the population decreases below some defined threshold value. This percentage-time estimate may be used as an index of risk analogous to that in Approach 2 above.

Selection of a Level 3 Approach--The choice of an approach to Level 3 risk characterization will depend on site-specific conditions and the availability of data on resource populations and their responses to contamination. Therefore, specific recommended approaches, their data requirements, and costs are not identified here. Development of ecotoxicological models is in its infancy, especially when extrapolating data on individuals to predict population behavior (Levin et al. 1984). In most cases of Puget Sound dredged material disposal analysis, data limitations will preclude the application of Approaches 2-4 above. At present, the U.S. COE (1985) testing strategy for dredged material would only permit the application of Approach 1, which still depends on a verifiable dynamic model of transport and fate. Nevertheless, the usefulness of more complex risk analyses approaches is being demonstrated by the Field Verification Program in Long Island Sound (Gentile et al. 1982, 1983a,b; Bierman et al. 1986).

COMPARATIVE ANALYSIS MODEL

The purpose of this section is to describe a model for integrating information on ecological and human health risk estimates to evaluate various options for dredged material disposal. The model is a framework for aggregating variables with diverse units into a common measure of relative risk used to rank disposal options. The comparative analysis model recommended by workshop participants was the Multi-Attribute Tradeoff System (MATS) described by Brown and Valenti (1983). The categories of variables to be considered, the format for display of the data, and the basic elements of the MATS model are discussed in the sections below.

Comparative evaluation of risk posed by different disposal options is essentially a risk-benefit analysis. Factors to be considered in the evaluation of each disposal option include:

- Risk estimates
- Economic efficiency
- Equity (e.g., spreading of risk among different segments of the population)
- Administrative feasibility
- Public acceptability.

Not all of these factors can be judged in an objective quantitative manner. The comparative analysis model described below focuses specifically on risk estimation, although cost factors are addressed in a general way. A complete consideration of cost and other factors just listed are beyond

the present scope of work. For a more comprehensive treatment of plan evaluation and decision-making approaches, consult Keeney and Raiffa (1976), Brown et al. (1980), and Edwards and Newman (1982).

Variables and Display of Data

Risk estimates and other data for evaluation of disposal options should be displayed in a concise format that facilitates comparative analysis. The recommended format is a matrix of variables by disposal option, as shown in Table 6. The primary categories of data are:

- Dredged material volume
- Cost of dredging, disposal, and monitoring
- Sediment chemistry
- Bioaccumulation
- Toxicity bioassay
- Human health risk.

Note that data on various fractions of dredged material (e.g., elutriate, effluent, leachate) may be presented separately under the category of sediment chemistry.

Based on the individual variables, integrative variables (or rank scores) are:

- Net hazard rank
- Net cost rank
- Hazard to cost ratio.

The net hazard rank is the score derived from the MATS model, as explained below. Methods of cost analysis and net cost rank scores are not discussed. However, the general approach used in MATS could also be applied to analysis of cost-effectiveness.

Elements of the Multi-Attribute Tradeoff System (MATS)

The key elements of MATS are selection of specific factors (variables in Table 6), creation of comparative-risk scales and functions for different factors, and assignment of relative weights to different factors. Specific factors may vary with initial results of dredged material testing. For example, the selection of contaminants of concern is based on their relative concentrations measured in the dredged material, as well as their relative toxicity (see above, Hazard Identification). Measures of risk based on bioassays or human health assessment models will depend partly on complexity of the transport and fate model used to predict exposure concentrations in the environment (see above, Exposure Assessment).

TABLE 6. RISK MANAGEMENT MATRIX FOR SEDIMENT DISPOSAL OPTIONS

VARIABLE	UNITS	UNCONFINED AQUATIC		CONFINED AQUATIC		UPLAND	
		ZONE A1	ZONE A2	ZONE C1	ZONE C2	ZONE U1	ZONE U2
DREDGED MATERIAL VOLUME	M^3						
COST: DREDGING	$						
DISPOSAL	$						
MONITORING	$						
SEDIMENT CHEMISTRY							
PCBs	MG/KG DRY						
METALS	MG/KG DRY						
BIOACCUMULATION							
FIELD	MG/KG WET						
LAB	MG/KG WET						
TOXICITY BIOASSAY							
TEST 1	% RESPONSE						
TEST 2	% RESPONSE						
HUMAN HEALTH RISK							
ROUTE 1	RISK						
ROUTE 2	RISK						
TOTAL	RISK						
NET HAZARD RANK	RELATIVE						
NET COST RANK	RELATIVE						
HAZARD : COST RATIO	RELATIVE						

The comparative-risk scales discussed herein correspond to the constant-worth scales of Brown and Valenti (1983), where risk is interpreted as an inverse of worth. A constant-worth or comparative-risk scale has equal value or importance for each interval on the scale. The scale for the original variable may or may not be a comparative-risk scale. For example, one index of risk for a chemical variable may be Elevation Above Reference (EAR):

$$EAR_{ij} = \frac{CS_i}{CR_{ij}} \qquad (18)$$

where:

EAR_{ij} = Elevation Above Reference value for chemical i and disposal option j, where i and j correspond to row and column coordinates within the Sediment Chemistry category of Table 6

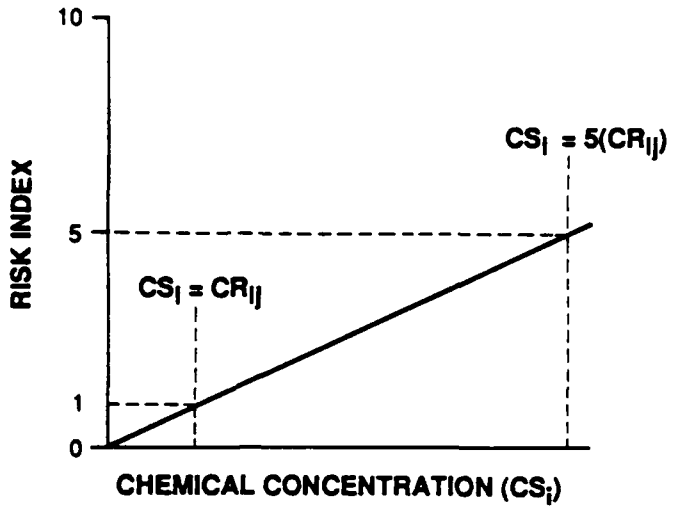
CS_i = Concentration of chemical i in dredged material

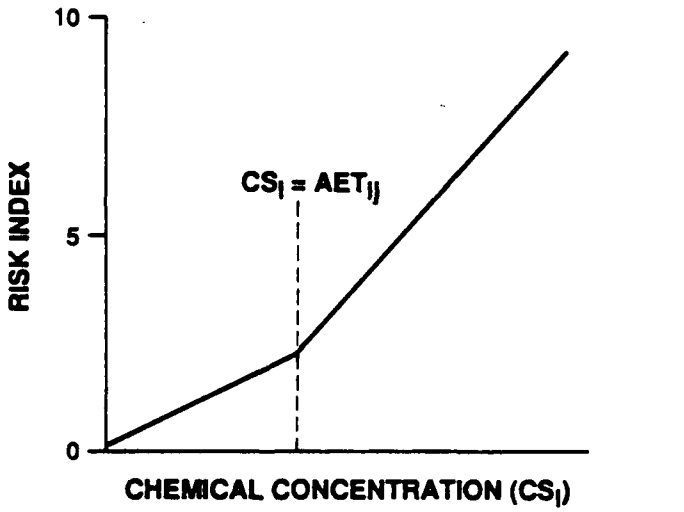
CR_{ij} = Concentration of chemical i in sediment from a reference site.

When the original scale of chemical concentration is itself a comparative-risk scale, a plot of relative risk vs. concentration is a straight line (e.g., upper panel of Figure 9). However, one unit increase of a chemical within a range of low concentrations may not represent the same risk increment as one unit increase within a range of high concentrations. For example, concentration increases above an Apparent Effect Threshold (AET) (Tetra Tech 1985b) may imply greater risk than those below the AET. In this case, the slope of the constant-risk function relating the EAR to the chemical concentration would increase above the AET (lower panel of Figure 9). Normalizing the values of site-variables to reference values creates comparative-risk scales. In most cases, it may be sufficient to use EAR directly as risk indices, implying a simple proportional relationship between risk and chemical concentration. Note that MATS requires the user to define the range of interest for the "impact scale" (e.g., chemical concentration scale in Figure 9). The risk-index scale is then normalized to a range of 0-1.0. A non-normalized scale is shown in Figure 9 only to illustrate the relationship between the risk index and the EAR.

In providing the flexibility to use nonlinear comparative-risk functions, MATS also allows the risk analyst to incorporate shifts in perceived risk based on professional judgment, agency policy, or public perception. The judgments and policies that are implicit in evaluations of dredged material disposal options can be made explicitly by defining appropriate comparable-risk functions within MATS.

The next key element of MATS is the judgment of the relative importance of the risk variables. The analyst defines the relative importance of variables by assigning a set of factor weights. Factor weights are expressed as decimal fractions that sum to 1.0 across all factors (Brown and Valenti 1983). These weights are essentially coefficients that serve as multipliers for each risk index, converting the comparable-risk scale of Figure 9 into a weighted-risk scale. For example, human health risk might be weighted more heavily than toxicity bioassay risk by policy decision. Consequently,





NOTE: In upper panel, the risk index is equal to the Elevation Above Reference value

CR_{ij} = Chemical concentration in reference area for option $_j$

CS_i = Chemical concentration in dredged material

AET_{ij} = Apparent Effect Threshold for chemical $_i$ and option $_j$

Figure 9. Hypothetical comparative-risk functions for a sediment chemistry variable.

the human health risk index might be given a factor weight of 0.75, where the amphipod toxicity index might be assigned a weight of 0.25 (assuming only these two risk factors are involved).

The final step in the MATS evaluation of disposal options is to determine the rank score (termed Net Hazard Rank in Table 6) for each disposal option. MATS performs this function by summing weighted risk indices across factors to obtain the rank score (i.e., sum of weighted indices within each column of Table 6). The analyst can then compare disposal options using the rank score as an integrated index of risk.

UNCERTAINTY ANALYSIS

Risk assessments are always based on limited data, assumptions, and models. Because of the limitations inherent in such assessments, it is essential to discuss the uncertainties associated with estimates of exposure, toxicological hazard, and risk. Methods for uncertainty analysis are discussed by Cox and Baybutt (1981), Morgan (1984), and Whitmore (1985). The U.S. EPA guidelines on exposure assessment (U.S. EPA 1986b) describe general approaches to characterization of uncertainty.

The purpose of this section is to:

- Identify generic categories of uncertainty in risk analysis of dredged material disposal options
- Present general approaches to uncertainty analysis
- Provide examples of uncertainty analysis from the workshop and from the literature.

The examples presented below illustrate the range of approaches to uncertainty analysis that may be applied to the models discussed earlier.

Categories of Uncertainty

Uncertainty is inherent in each stage of a risk analysis, from the initial characterization of dredged material composition to the final calculation of risk. General categories of uncertainty are:

- Variance and bias in estimating a variable measured during sediment testing:
  - Representativeness and variability of sampling
  - Uncertainty in analytical chemistry or bioassay test measurements
- Variance and bias in estimating a calculated variable due to corresponding variance and bias in component variables, for example:

- Variance in exposure estimate due to uncertainty in estimate of media contact rate and concentrations of contaminants

- Variance in carcinogenic risk estimate due to uncertainty in estimates of exposure and carcinogenic potency factors

- Model uncertainty; for example, dose-response and exposure models, MATS:

  - Variables considered in model

  - Statistical distributions of variables

  - Functions

  - Variable weights and scaling (MATS).

Approaches to Uncertainty Analysis

Different kinds of uncertainty may require different analytical strategies for characterization of uncertainty. For example, calculating a 95 percent confidence interval for an estimated mean value would be appropriate for a normally distributed variable. For a variable exhibiting some other distribution, the range of observations may be a more appropriate expression of uncertainty. Model uncertainty must be addressed by evaluating alternative model formulations.

Approaches to treatment of uncertainty in model coefficients used in risk analysis include (Morgan 1984):

- Perform analysis using **single-value-best-estimates** for model coefficients, **without uncertainty analysis**

- Perform **single-value-best-estimate** analysis, **with sensitivity calculations** and appropriate discussion of uncertainty

- Estimate some **measure of uncertainty** (e.g., standard deviation) for each model coefficient and use **error propagation** methods to estimate uncertainty of final exposure or risk value

- Characterize subjectively the **probability distribution** of each model coefficient and **propagate error** through stochastic simulation

- Characterize important model coefficients using a **parametric model**, and perform risk analysis using various **plausible values** of each of the coefficients

- Determine upper and lower **bounds** on model coefficients to yield **order-of-magnitude estimates** and range of possible answers.

Morgan (1984) refers to the first two approaches as "single-value-best-estimate analysis," to the second two as "probabilistic analysis," and to the final two as "parametric/bounding analysis." The analytical strategies are listed above in approximate order, indicating the amount of uncertainty in the model coefficients. Single-value-best-estimate analysis is appropriate when model coefficients are precisely known. Bounding analysis is most appropriate when little is known about the values of the coefficients. Finally, note that the techniques listed above do not address model uncertainty, which must be handled by exploratory examination of outcomes based on alternative equations.

Examples of Uncertainty Analysis

Order-of-Magnitude Bounding Analysis--

During the risk analysis workshop, Dr. Curtis Brown recommended Range-Estimating Program for initial analysis of uncertainty. The program user specifies a model, the key variables, and means and ranges of variables. Probability distributions may also be specified for each input variable. The program performs a Monte Carlo simulation (1,000 scenarios) to estimate probability distribution of outcomes. An example of the use of the Range Estimating Program is provided in Appendix D.

Dr. Alan Ehrlich provided the workshop with some possible ranges of uncertainty in calculating the carcinogenic potency estimates used in human health risk assessment. As shown in Table 7, each key factor in the dose-response has an estimated range of uncertainty. For example, the variation in estimates of carcinogenic potency in humans based on extrapolation from bioassay animals on a body weight basis vs. a surface area basis is a factor of 2-12. Use of an upper bound of the 95 percent confidence limit (UCL) as the potency value may result in risks estimates 2-3 times higher than those based on a maximum likelihood estimate (MLE) of potency. Other choices in selection of assumptions or data involving uncertainty are 1) the use of data for malignant tumors only vs. data for malignant plus benign tumors, 2) the use of data for the average species vs. the most sensitive species, 3) the use of an administered dose vs. an estimated tissue-dose based on pharmacokinetic modeling, and 4) nonequilibrium buildup of carcinogen in tissue vs. equilibrium kinetics. The range in each factor may lead to risk estimates differing by as little as a factor of 16 or as much as a factor of 10,800.

Probability Distributions for Risk--

Crouch et al. (1983) characterized the uncertainty in human health risk estimates using a linear no-threshold, dose-response model similar to the one used by U.S. EPA (see Equation 6 above). Instead of using a 95 percent upper confidence limit on the parameter B_{ij} as an estimate of carcinogenic potency, as U.S. EPA (1980b) does, Crouch et al. (1983) modeled the uncertainty in B_{ij} as a lognormal probability distribution. Combining lognormal probability distributions for exposure and for an interspecies extrapolation factor with the probability distribution for potency (B_{ij}) Crouch et al. (1983) derived:

TABLE 7. CHARACTERIZATION OF UNCERTAINTY RANGES FOR SELECTED FACTORS IN CARCINOGEN RISK ASSESSMENT

Factor	Choice of Approach	Uncertainty Range[a]
Interspecies extrapolation	Body weight vs. surface area basis	2 12
Potency estimate	Maximum likelihood estimate vs. upper confidence limit	2 3
Tumor type	Malignant vs. (Malignant + Benign)	2
Sensitivity	Average animal vs. sensitive animal	2 5
Dose expression	Pharmacodynamics vs. effective dose	6
Dose kinetics	Non equilibrium build up vs. steady state	2 5
	Total	6 10,800

[a] See text for explanation.

Reference: Ehrlich (16 December 1985, workshop presentation

- Lognormal probability distribution for excess cancer risk
- Median, mean, and 98th percentile estimates of risk
- Variance of the mean of lognormal transformation of risk, which is calculated as the sum of the variances of the log transformed potency factor, the interspecies extrapolation factor, and the exposure dose.

An example of the distribution of risk calculated for 100 ug/L trichloroethene in drinking water is shown in Figure 10.

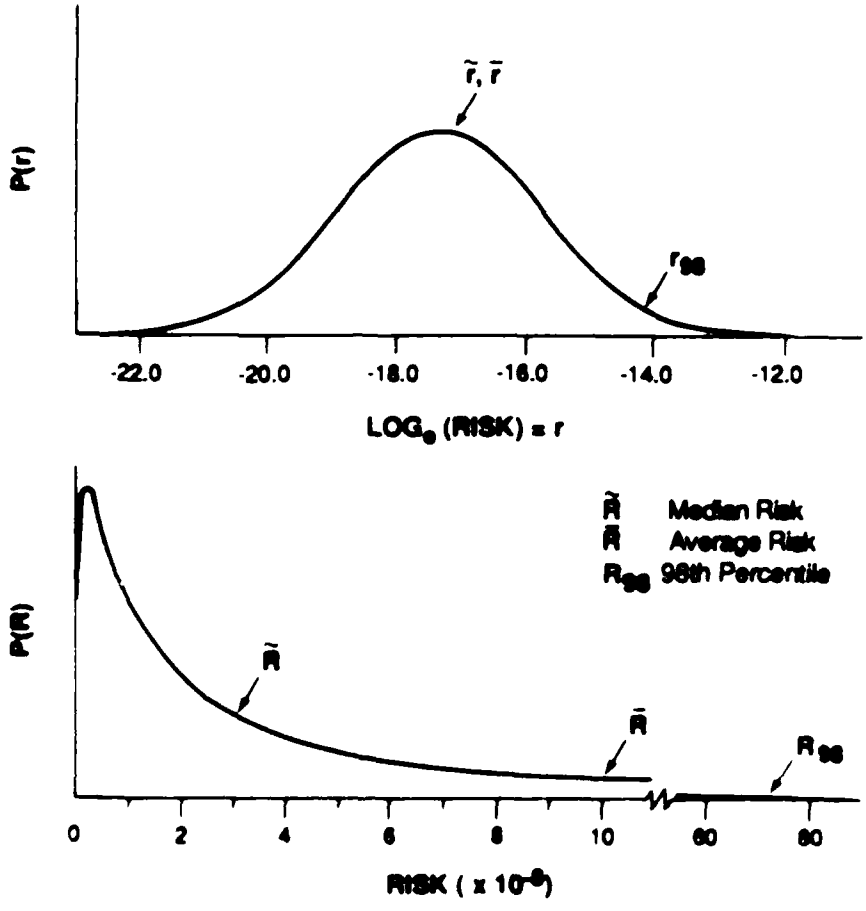
Model Uncertainty

A great deal of effort has been expended by risk analysts exploring alternative low-dose extrapolation models. These models are used to extrapolate dose-response data from high doses used in bioassays to low doses of interest for human health risk assessment. The linearized multistage model (U.S. EPA 1980b) is one of a class of statistical quantal response models. As shown by Dr. Alan Ehrlich and Dr. David Eaton at the workshop, a wide range of risk estimates (as much as [illegible] to seven orders of magnitude) may be obtained, depending on the model selected for low-dose extrapolation. An example of the divergence of risk estimates among models is shown in Figure 11. Discrimination among models by statistical goodness-of-fit criteria is impossible. Therefore, the choice of a model must be based on scientific judgment of biological realism and policy. The multistage model was selected by U.S. EPA (1980b, 1986a) as the most appropriate model for human health risk assessment in the absence of data on alternative mechanisms of carcinogenesis.

[illegible]

[illegible]

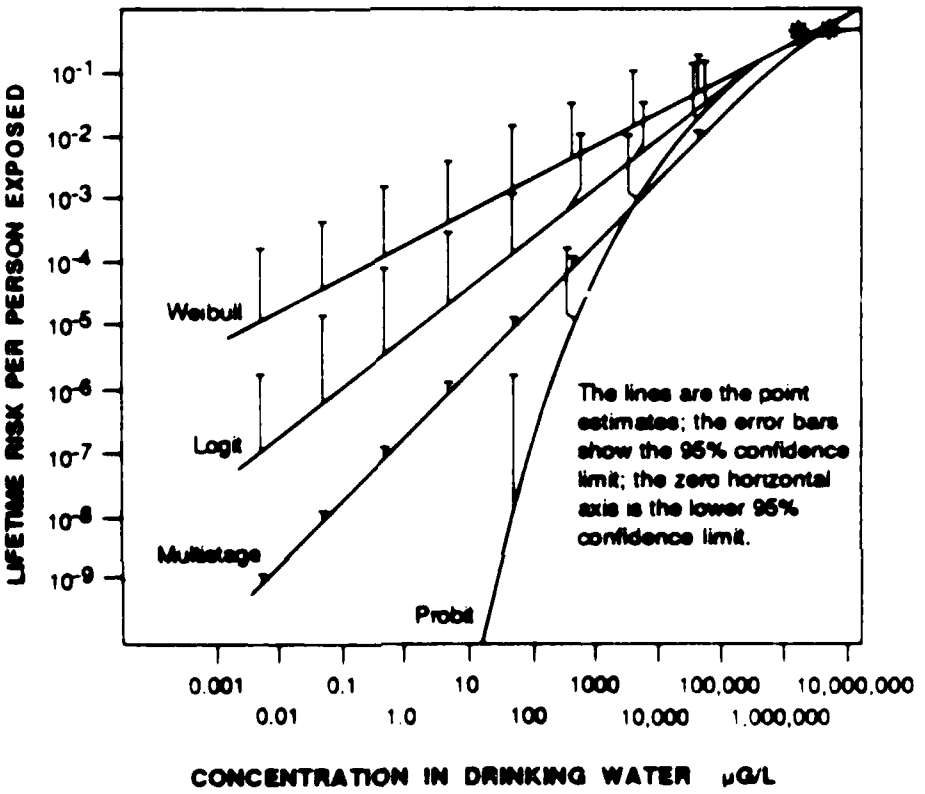
- [illegible]
- [illegible]



Note P(r) is the probability of a given log-transformed value (r) of risk (R)
P(R) is the probability of a given value of risk (R).

Reference Crouch et al. (1983)

Figure 10. Lognormal risk distribution (average annual cancer risk from 100 ug/L trichloroethene in drinking water).



Bioassay data are the starred points in the upper right hand corner. The lifetime risk per person exposed was calculated assuming that the risk due to inhalation and dermal exposure is equal to that due to ingestion exposure. The error introduced by this simplifying assumption is less than the widths of the lines shown.

Through ingestion, inhalation, and dermal exposure, assuming that ingestion equals inhalation plus dermal.

Reference: Cothern et al. (1986)

Figure 11. Bioassay data and model extrapolations for exposure to trichloroethene through ingestion.

- Derive the RfD by application of an interspecies extrapolation factor and an uncertainty factor to account for variability that is not modeled.

An example of the application of the approach is given by Dourson et al. (1985).

Both model uncertainty and variable uncertainty may be addressed qualitatively by examination of assumptions. As Dr. David Eaton pointed out at the workshop, key assumptions used in exposure assessment greatly influence final risk estimates. Some of these assumptions are dependent on samples taken to characterize exposure concentrations (e.g., the validity of the analytical procedure, the degree to which the samples represent the real environment, and the assumed bioavailability of measured contaminants). Actual bioavailability depends on matrix effects, chemical speciation, and route of exposure. Other assumptions are sample-independent [e.g., characterization of the exposed population, the validity of using standard exposure coefficients (e.g., ingestion rates, inhalation rates), and the assumed frequency and duration of exposure]. Dr. Eaton pointed out the major pitfalls of quantitative health risk assessment:

- Failure to document all pertinent assumptions
- Use of single-value-best estimate analysis, without uncertainty analysis
- Misrepresentation of the exposed population by implying exposure of the population-at-large
- Failure to consider probability of exposure, where appropriate
- Failure to translate results into a form understandable by the general public and to provide perspective by using relative risk comparisons.

The importance of model documentation and analysis of assumptions was stressed repeatedly by workshop participants. Example formats for display of integrated exposure analysis (Table 7 above) and assumptions (Table A-2 in Appendix A) suggested in this report provide a framework for documentation and analysis of assumptions.

EXAMPLE APPLICATION OF COMPARATIVE RISK ANALYSIS

An example risk analysis of dredged material disposal options is outlined below. The source of sediments is assumed to be the lower Duwamish River, Seattle, WA. Three disposal options are considered:

- Deepwater, unconfined disposal site with the Fourmile Rock Disposal Site in Elliott Bay as an example
- Nearshore, confined disposal site with the Terminal 90 91 complex in Elliott Bay as an example
- Upland sanitary landfill disposal site with the Midway Landfill as an example.

Although the site characterizations included in this analysis are based on the actual sites listed above, the test data used to characterize the dredged material are hypothetical. Each stage of a risk assessment is illustrated for the deepwater, unconfined disposal option. For other potential disposal sites, hypothetical risk estimates are assumed. Risk indices for all three disposal options are evaluated and integrated with the comparative analysis model, the Multi Attribute Tradeoff System (MATS).

The example below illustrates the steps of a comparative risk analysis, using a Level 2 exposure analysis and risk characterization. The Level 2 analysis was selected to show the major elements of a quantitative risk assessment that may form the framework for future evaluation of dredged material disposal options. An example based on qualitative risk assessment (Level 1) would not have demonstrated the quantitative modeling framework. At the other extreme, a Level 3 analysis incorporating temporal and spatial variation of contaminant concentrations was not necessary to illustrate the key aspects of the approach.

After the initial site characterization, the example is presented as an outline of steps and sample calculations performed in comparative risk analysis. Final results are expressed as a ranking of disposal options in terms of relative risk. In an actual application of the approach, supporting text would include interpretation and discussion of data, assumptions, and results during each stage of the analysis. In actual practice, quantitative uncertainty analysis is recommended where appropriate. Examples of uncertainty analysis were presented earlier (see above, Conceptual Approach to Risk Analysis of Dredged Material Disposal Options) and do not need to be repeated below. Single value best estimate analysis and parametric bounding approaches would use the same framework as specified in the example. Application of sensitivity analysis or parametric bounding approaches would involve repeating, for different sets of assumptions or input values, the exact steps shown below. Probabilistic analysis such as error propagation would involve some additional statistical characterization of model variables, but the basic models would remain unchanged.

SITE CHARACTERIZATIONS

The primary study area, including Elliott Bay and the lower Duwamish River, is shown in Figure 12. This area includes the source of dredged material and two aquatic disposal sites. The upland site, Midway Landfill, is located just south of the area shown in Figure 12.

Sources of Sediments: East, West, and Duwamish Waterways

Currently, a plan to conduct extensive dredging in the Duwamish, East, and West Waterways as part of a federal navigation improvement program includes the following dredging activities:

- Increase the channel depth in the East and West Waterways from 34 to 39 ft
- Increase the channel width and depth of the Duwamish Waterway (downstream of the First Avenue South Bridge) from 200 ft by 30 ft to 250 ft by 39 ft
- Deepen Turning Basin No. 1 at the southern end of Harbor Island (U.S. COE 1983).

Land use in the lower Duwamish River area is almost exclusively industrial. The shoreline is nearly completely altered by riprap, bulkheads, or overwater development (i.e., pilings). The entire East Waterway shoreline is owned or occupied by the Port of Seattle and is used for vessel loading and cargo storage and transport. The western shore of the East Waterway is dominated by the Port of Seattle's Terminal 5, whereas the western shore is occupied by two large shipyards, a tank farm, and several industrial/manufacturing facilities. The Duwamish River shoreline north of the First Avenue South Bridge (with the exception of Kellogg Island) is occupied predominantly by a shipyard and Port of Seattle facilities.

Sediment in the navigation channel is composed of organic detritus, river sand and silt, and coal fragments. Considerable amounts of wood and plant fibers are intermixed with the sediments, increasing in quantity downstream. As a result of heavy industrial activity in the area, sediments in the lower Duwamish River have been contaminated by a variety of metals and chlorinated organic compounds (e.g., PCBs). Surface sediments also contain oil, tar, and other petroleum products and their derivatives.

Deepwater, Unconfined: Fourmile Rock Disposal Site

The Fourmile Rock Disposal Site is demarcated as a 900 ft radius around a point defined by 122°25'00" longitude, 47°37'35" latitude (Figure 12). This disposal area is located near the northwest entrance to Elliott Bay on an offshore slope ranging in depth from 300 ft to 550 ft. The majority of the sediment disposed of at the Fourmile Rock Disposal Site has come from the Duwamish Waterway. Other sources of sediment have included substantial quantities originating from the Bremerton U.S. Navy facilities, Lake Washington Ship Canal, Eagle Harbor, and Rich Passage (Seattle Department of Construction and Land Use 1984a,b).

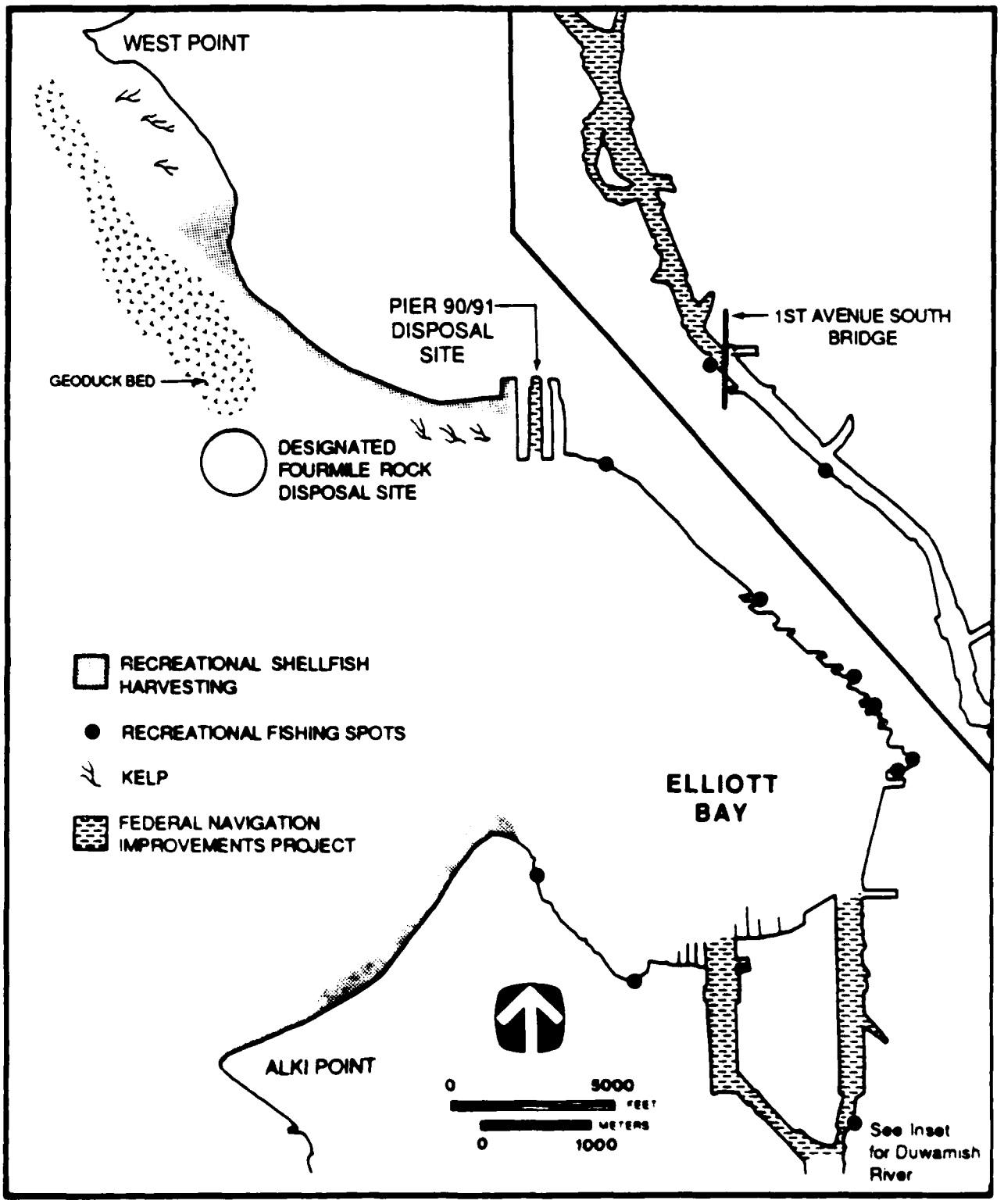


Figure 12. Key resources and sediment management locations in Elliott Bay.

Land use from West Point toward Seattle downstream includes:

- Discovery Park - a heavily used park with nearly 1 mi of beach and intertidal public access
- Single family residences - both along the beach and on the bluff for approximately 2 mi of shoreline east of Discovery Park
- Smith Cove Park - a small waterfront park offering limited (approximately 1,000 ft) public access to a beach and intertidal area
- Port of Seattle Terminal 91 - comprised of Piers 90 and 91 and upland storage and warehouse areas
- Myrtle Edwards Park - borders Port of Seattle property along the shoreline from Terminal 91 to Pier 71 (about 1.5 mi) and contains a public fishing pier at Terminal 89. This park affords little to no public access to beaches or intertidal areas, as the shoreline is 100 percent riprapped.

Elliott Bay supports a wide variety of biological resources including a substantial supply of commercially and recreationally harvested fish, shellfish, and algae. A portion of the migration route for chinook, coho, and chum salmon, steelhead and cutthroat trout, and dolly varden roughly follows the northern shoreline of Elliott Bay in the vicinity of the disposal site. The benthic assemblages (polychaetes, mollusks, and crustaceans) on and in the vicinity of the disposal site may serve as a food source to certain species of bottomfish. In addition, it is possible that the disposal site and surrounding area is used by English sole as spawning grounds (Stober and Pierson 1984). Other biological resources of commercial or recreational importance near the disposal area are shown in Figure 12, including:

- A geoduck bed northwest of the site
- Kelp beds off Discovery Park and Smith Cove Park
- Intertidal and subtidal shellfish habitat from Discovery Park to Terminal 91 (Discovery Park beaches are closed to shellfish harvesting, but private use of Magnolia Beach for shellfish harvesting is common)
- Several species of crab, shrimp, and squid that are commonly found in Elliott Bay.

Marine mammals that have been observed in Elliott Bay include Steller sea lions, harbor seals, and killer whales.

Nearshore: Piers 90 and 91--

The intertidal, shallow subtidal, and deepwater area between Piers 90 and 91 in Elliott Bay is a proposed disposal site for dredged material.

The disposal area is located on Port of Seattle property, east of Smith Cove Park on the northern shore of Elliott Bay (Figure 12). Shoreline use in the immediate vicinity includes single family residential property (predominately in the Magnolia area). Port of Seattle upland storage and warehouse facilities, and two waterfront parks bordering the site (Smith Cove Park and Myrtle Edwards Park). With the exception of the Magnolia neighborhood area, the surrounding uplands (Interbay and Elliott Ave. corridor) are dominated by industrial and commercial land uses.

Public access to the shoreline is afforded almost exclusively by the two parks. Smith Cove Park occupies about 1,000 ft of shoreline west of Piers 90 and 91 and provides public access to a beach and intertidal area which can be used for fishing as well as the recreational gathering of shellfish and algae. Myrtle Edwards Park is a thin strip (100-300 ft) of riprapped shoreline extending about 1.5 mi from Piers 90-91 to Pier 71. Myrtle Edwards Park includes a popular public fishing pier at Terminal 89, approximately 1,000 ft from the site.

The disposal site between Piers 90 and 91 is approximately 500 ft wide by 2,500 ft long with an average water depth of 38 ft. The site is currently comprised of 4 ac of intertidal/shallow subtidal habitat and 27 ac of deepwater habitat. The site supports a wide variety of biological resources, including at least 19 species of fish having recreational or commercial value. Juvenile chum and chinook salmon, rock sole, English sole, Pacific cod, and sand sole use the area for feeding (U.S. COE 1983). In addition, juvenile salmon probably use the area under the piers as shelter from predators. Other resources provided by habitats on the site include:

- Sedimentary Intertidal Habitat - provides benthic macroinvertebrates that are preyed upon by shore birds (e.g., dowitches, killdeers, yellowlegs, and sandpipers)

- Structural Intertidal Habitat - provided by pilings, riprap, and bulkheads supports a variety of invertebrates (e.g., mussels, barnacles, and limpets), and algae

- Deepwater Habitat - the area between and underneath the piers provides habitat for marine birds (e.g., gulls, connorants, grekes, and coots).

Upland Disposal: Midway Landfill

The Midway Landfill site, located at 24800 Pacific Highway South in Kent, WA, was operated by the City of Seattle from 1966 to October, 1983. Land use adjacent to the site ranges from single family residential to commercial. The entire area east of the site (across Interstate 5) and south of the site is occupied by single family homes. There is a small neighborhood park at 248th Street, across the freeway from the site. On the north side, the site is bordered by vacant land (along Interstate 5), commercially zoned land (along Route 99), and a mobile home park (about 100 ft north of the site). To the east, the site is adjacent to commercial property (along Route 99), and multi unit residential property and a mobile home park to the southeast. Parkside Elementary School is located approximately 1,500 ft west of the site in a predominantly single family residential

area. The landfill is approximately 60 ac in area and is up to 120 ft deep. The predominant soil type underlying the landfill is composed of advance glacial outwash deposits over consolidated sand and gravel with minor amounts of silt. These deposits were quarried prior to the use of the site as a landfill. There are occasional silty units within the outwash that may create localized zones of perked groundwater (Seattle Engineering Department 1985). The site's highest elevation is 409 ft at the southeast corner, where it gently slopes to the northwest and west.

Midway Landfill is located near or on a regional groundwater divide separating groundwater that flows eastward toward the Green River and westward toward Puget Sound. Although the direction of groundwater flow is unknown in the vicinity of the site, it is surmised to flow south. There are three public water supply wells in the vicinity of the site; the nearest two are 2 mi away to the northwest. It is unknown if any of these wells are draining water from the same glacial outwash deposits that exist in the vicinity of the site. Available data indicate that groundwater in underlying outwash is probably below the level of landfill material (i.e., 276 ft elevation for water; 280-290 ft elevation of lowest landfill material (Seattle Engineering Department 1985). All surface water originating on the site is contained onsite in retention ponds where it is periodically removed and disposed of by the City of Seattle.

Given the nature of the site, there are no biological resources of note. However, the site is directly used by several species of avian and mammalian scavengers (e.g., gulls, crows, mice, and rats). Biological resources near the site include a second growth stand of mixed conifer and hardwood forest (beyond the site's northern boundary); a 10 ac wetland near Parkside Elementary School (approximately 800 ft west of the site); and the North Fork of Smith Creek (headwaters are the aforementioned wetland). The stand of forest harbors a variety of urban wildlife including several bird species (e.g., sparrows, starlings, juncos, chickadees) and mammal species (most likely mice, voles, and squirrels). The wetland area harbors a variety of plant life including water parsley, skunk cabbage, creeping buttercups and sedges, with an overstory of western red cedar, red alder, and black cottonwood (Seattle Engineering Department 1985). Mallards, wood ducks, scamp, and Canadian geese also frequent the vicinity of Smith Creek and associated wetlands. Smith Creek provides spawning and rearing habitat for coho and chum salmon, and possibly cutthroat trout. There are no data on the fisheries resources of this waterway.

HAZARD IDENTIFICATION

Characterize Dredged Material

- Estimate contaminant concentrations in bulk dredged material and reference sediment (see hypothetical data in Appendix F, Table F-1). In this step and subsequent ones, sediment B is selected to represent dredged material from the Lower Duwamish River.

- Estimate contaminant concentrations in elutriate, pore water, surface runoff, and leachate from dredged material and reference water (see hypothetical data in Appendix F,

Table E-2 for leachate. Only data for leachate are used in the example below).

Select Contaminants of Concern

- Identify contaminants of concern
- For simplicity in this example, PCBs and mercury (Hg) are considered as the only contaminants of concern.

Compile Toxicity Profiles (PCBs, Hg)

- Key physical chemical properties (Table B-
- Evaluation of carcinogenic potential (Table B-
- Concise summary of toxicological effects data as presented by Patken and Lewis (1983).
- An example toxicity profile for PCBs is presented below. For additional information, consult the following sources: by Fishbein (1974), IARC (1978), US EPA (1980a), and Jeffery (1983). For details on toxicity of PCBs to humans, see U.S. Department of Health and Human Services (1985

PCBs are composed of a biphenyl nucleus with varying degrees of chlorine substitution at the constituent carbons. There are 209 different mono substituted to hexa substituted PCBs that range from [illegible] per molecule. One of these (1980). For example, Aroclor 1221 contains 21 percent chlorine and Aroclor 1260 contains 60 percent chlorine. The degree of chlorination is important in determining the [illegible] of PCBs. The higher the chlorine content, the greater the potential for [illegible] and the greater the [illegible] of PCBs to [illegible] (Safe 1980). PCBs are characterized by [illegible] (1984). [illegible] (1984).

PCBs may be [illegible] in the [illegible] of PCBs [illegible] (1980).

TABLE 8. TOXICITY PROFILE FOR MERCURY AND PCBS[a]

Property	Mercury[b]	PCBs[c]
CAS Number	7439-97-6	1336-36-3
Physical-Chemical		
Molecular weight	200.6-318.7	154.2-498.7
Vapor Pressure (mm Hg)	0.012-0.028	2.8×10^{-9} - 7.6×10^{-5}
Solubility (mg/L)	0.056-400,000	<0.001-5.9
Log K_{ow}[d]	N/A[e]	4.0-6.9
Log Bioconcentration Factor[d]	2.0-4.6	1.9-5.2
Carcinogenic Status	Noncarcinogen	Probable human carcinogen[f] -- Sufficient animal evidence -- Inadequate human evidence
Acute Toxicity		
Human (mg/kg body wt) LD_{50}	29[g]	--
Mammal (mg/kg body wt) LD_{50}	1.0-40.9	1,010-16,000
Aquatic (mg/L) LC_{50}	0.015-32.0	0.001-61.0
Chronic Toxicological Effects		
Humans	Motor and sensory impairment leading to paralysis, loss of vision and hearing, and death. Kidney dysfunction.	Skin lesions, liver dysfunctions, and sensory neuropathy.
Mammals	Reproductive impairment and teratogenic effects.	Hepatotoxicity, fetotoxicity, skin lesions, and hepatocellular carcinoma.
Aquatic Organisms	Developmental and structural anomalies, suppression of growth and reproduction, impairment of behavior.	Reproductive and developmental impairment.

[a] This is an example toxicity profile and is not intended to be comprehensive.

[b] Mercury may occur in its elemental form, as inorganic salts, or as organic complexes. Consequently, the chemical and toxicological properties vary tremendously depending on the degree of complexation or metal speciation.

[c] Physical chemical properties and toxicity vary according to the degree of chlorine substitution, the number of adjacent unsubstituted carbons and steric configuration.

[d] Tetra Tech (1985a).

[e] N A - not applicable.

[f] U S EPA (1980b, 1984a, 1985a); IARC 1978.

[g] For mercury (II) chloride via oral route of exposure (Tatken and Lewis 1983).

depending on the experimental organisms. Once absorbed, PCBs are initially concentrated in the liver, blood, and muscle tissues. Long-term storage is in the skin and adipose tissue (Safe 1980).

PCB metabolism varies with the degree of chlorine substitution and the number of unsubstituted pairs of carbons on the biphenyl nucleus. Mono- to penta-substituted PCBs are the most easily metabolized, with tissue half-lifes in experimental animals of less than 7 days. However, hexa-substituted PCB is persistent and unlikely to ever be completely eliminated (Matthews and Anderson 1976). Metabolized PCBs are excreted in varying proportions in the urine and bile, depending again on the chlorination state of the parent molecule.

Polychlorinated biphenyls (PCBs) have low acute toxicity, but are of public health concern because they are environmentally persistent, are easily bioaccumulated, and cause numerous chronic toxic effects in humans and animals (Letz 1983). Epidemiologic investigations of consumption of PCB-contaminated rice oil in Japan in 1967 provide much of what is known about human health effects of PCBs (Kuratsune 1976, 1980). However, in most instances, epidemiologic studies documenting health effects of PCBs are confounded by a number of factors: small study populations, lack of accurate exposure data, presence of potentially harmful co-contaminants, and lack of control of other health variables (Letz 1983). Consequently, most information regarding PCB toxicity is derived from controlled animal studies.

Experiments designed to determine lethal concentrations of orally administered PCBs in rats have shown that PCBs have a relatively low acute toxicity. That is, death resulting from a single exposure requires a massive dose unlikely to occur under environmental circumstances. LD50 concentrations (i.e., the exposure dose necessary to kill 50 percent of the test rats) ranged from 1,010 mg/kg body weight for Aroclor 1254 to 16,000 mg/kg body weight for Aroclor 4465 (Tatken and Lewis 1983). For Aroclor 1260, the single rat oral LD50 is 1,315 mg/kg, which is about the same as that for [illegible] (Tatken and Lewis 1983).

Single large exposures or repeated chronic exposure to PCBs may have [illegible] long-term toxic effects. As indicated above, PCBs tend to be [illegible] and metabolized in the liver, and redistributed to skin and [illegible] tissues for long-term storage. Thus, hepatotoxicity and skin lesions [illegible] manifestations of PCB poisoning. Transplacental transport of [illegible] may result in fetotoxicity. However, PCBs have not been shown to [illegible]genic. PCB metabolism results in bioactivation of potentially [illegible] intermediates, which is consistent with the formation of hepato-[illegible]nomas in experimental rats. Consequently, PCBs were listed [illegible] human carcinogens by the U.S. EPA (Fishbein 1974; U.S. EPA [illegible] the International Agency on Research on Cancer (IARC 1978). [illegible] considers PCBs as **probable** human carcinogens.

DOSE-RESPONSE ASSESSMENT

Compile Toxicologic Indices for Human Health Effects

Mercury, Reference Dose (RfD)--

- RfD = 0.02 mg/day = 0.0003 $mg \cdot kg^{-1} \cdot day^{-1}$
- Route - oral.
- RfD based on human data and estimated lowest-observed-adverse-effect level and an uncertainty factor of 10. Higher doses have caused nervous disorders.

PCBs, Carcinogenic Potency Factor (B_{ij})--

- B_{ij} = 4.34 $kg \cdot day \cdot mg^{-1}$ (U.S. EPA 1985a)
- B_{ij} is a plausible upper limit (95 percent confidence limit) value.

Summarize Results of Sediment Toxicity Bioassays

All hypothetical bioassay results reported below are from 96-h acute, lethal bioassays using bulk dredged material. Additional data compiled from a review of the literature for target species and contaminants of concern (PCBs, mercury) would also be summarized during a full dose-response assessment. In the exposed population analysis (see below, Exposure Assessment), infaunal amphipods (Rhepoxynius abronius), English sole (Parophrys vetulus), and geoduck clams (Panope generosa) are identified as appropriate biological indicator species. However, to illustrate a full range of example risk calculations presented later, it is assumed that dose-response data for geoducks are unavailable. Example dose-response data are presented below.

Bulk Sediment: Amphipod (Rhepoxynius abronius)--

- Amphipod sediment-dilution series bioassay results (Appendix E, Table E-3). Additional replication would be used in practice.
- LC_{50} = 40 percent dredged material
  Upper 95 percent Confidence Limit = X percent
  Lower 95 percent Confidence Limit = Y percent.
- Regression equation for estimation of excess percent mortality (PB_r) from percent dredged material (C), as follows:

$$\text{Log}_{10}\ (PB_r) = a + b\ \text{Log}_{10}\ (C+1)$$

  where a and b are estimates of intercept and slope coefficients, respectively.

Bulk Sediment: English Sole (Parophrys vetulus)

- Similar series of dilution bioassays were conducted using juvenile English sole.

- LC_{50} = 55 percent dredged material
  Upper 95 percent Confidence Limit = x percent
  Lower 95 percent Confidence Limit = y percent.

- For juvenile English sole:

$$\text{Log}_{10}\ (PB_r) = c + d\ \text{Log}_{10}\ (C+1)$$

EXPOSURE ASSESSMENT

Select Exposure Scenarios

- Potential pathways and routes of exposure:

  1. Water-column transport with dermal contact or ingestion of water and suspended sediments

  2. Sediment contact (on-shore) with dermal contact or ingestion of sediment

  3. Sediment contact (on-site) with dermal contact or ingestion of sediment (nonhuman species only)

  4. Food chain transfer of contaminants with ingestion of contaminants in food.

- Key exposure pathways and routes for chronic biological effects are Scenarios 2-4 above. Exposure Scenario 1 above is important only during the initial deposition of dredged material, except where resuspension of bottom sediments is a chronic problem. The potential for chronic resuspension of dredged material from the bottom can be predicted from data on currents near the bottom and calculation of potential sheer forces relative to grain-size composition of the dredged material. For this example, Scenario 1 above was determined to be negligible.

- Exposure Scenarios 2 and 3 are combined for nonhuman species, with exposure concentrations and probabilities treated below as functions of distance from the center of the disposal site.

Estimate Environmental Concentrations

Determine Extent of Transport/Fate Analysis (Level 1, 2, or 3 based on Figure 2 above)--

- Concentrations of PCBs and mercury in bulk dredged material exceed those for the reference sediment (Sediment B in

Appendix E, Table E-1). Concentrations of PCBs and mercury in bulk dredged material each exceed the respective maximum of the range of concentrations reported by Tetra Tech (1985b) for all Puget Sound reference areas.

- Dredged material volume is less than "x" (Figure 2 above).
- Mixing zone calculations based on methods of U.S. COE (1985) show area of deposition is less than "y" (Figure 2 above).
- Concentrations of PCB and mercury in bulk sediment and bioassay tissue are less than "guidelines" (Figure 2 above).
- Level 2 analysis of transport and fate is selected.

Determine Transport and Fate of Sediments (Figure 13)--

- Determine dredged material volume according to standard U.S. COE procedures.
- In Level 2 analysis, the model of Brandsma and Divoky (1976) as modified by Johnson (3 March 1986, personal communication) is used to predict the initial deposition of dredged material on the bottom. For simplicity, the pattern of deposition shown in Figure 13 is used to represent the model results. [Model results are normally expressed on a grid of cells, each having specified dimensions (Brandsma and Divoky 1976). The form of the model results (grid vs. concentric donut-shaped areas) does not affect application of the models in subsequent steps].
- Note that model simulations yield different spatial distributions and deposit thicknesses for different grain-size fractions. The upper panel of Figure 13 is a plot of the total distribution of bulk dredged material.

Determine Average Contaminant Concentrations in Bottom Sediments for Specific Areas (Figure 13)--

Average concentration of contaminant i in area x is calculated from Equations 1 and 2 for Area 1 (x=1, inner area), Area 2 (x=2, middle area), and Area 3 (x=3, outer area) as shown below. Depth of sediment mixing by bioturbation is assumed to be 5 cm.

- Without sediment mixing (Equation 2):

$$C_{PCB}(x) = CS_{PCB} = 2.000 \text{ mg/kg for all Areas 1-3}$$

$$C_{Hg}(x) = CS_{Hg} = 0.50 \text{ mg/kg for all Areas 1-3}$$

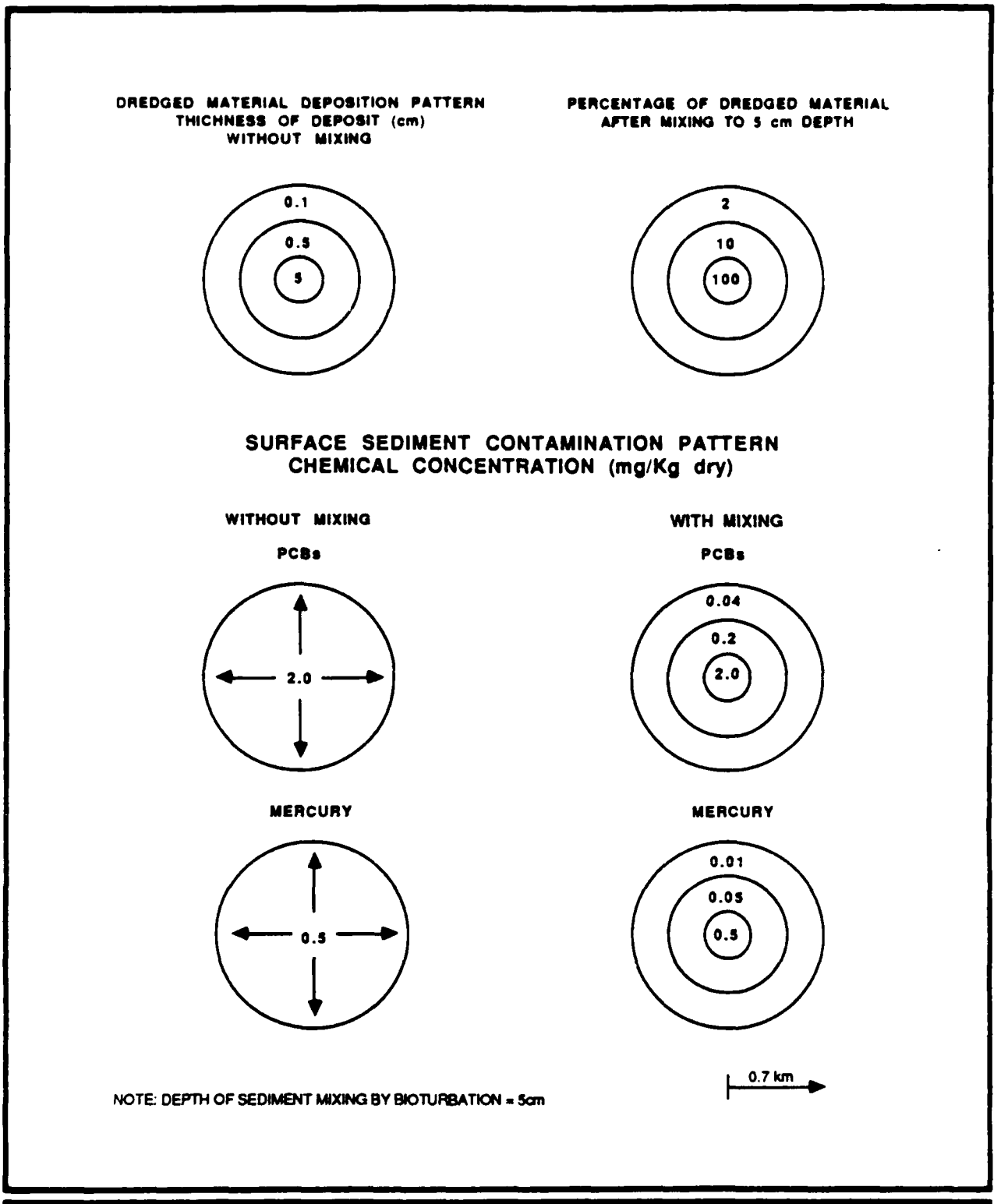


Figure 13. Hypothetical pattern of dredged material deposition and predicted contaminant concentrations.

- With sediment mixing (Equation 1):

  C_{PCB} (1) = 2.000 mg/kg

  C_{PCB} (2) = 0.200 mg/kg

  C_{PCB} (3) = 0.040 mg/kg

  C_{Hg} (1) = 0.50 mg/kg

  C_{Hg} (2) = 0.05 mg/kg

  C_{Hg} (3) = 0.01 mg/kg

Determine Average Contaminant Concentration in Bottom Sediments for Entire Disposal Site (Equation 3)--

- Without sediment mixing:

  C_{PCB} = 2.000 mg/kg

  C_{Hg} = 0.50 mg/kg

- With sediment mixing:

  Area 1 = A(1) = 0.03 km^2

  Area 2 = A(2) = 0.16 km^2

  Area 3 = A(3) = 1.30 km^2

  $\bar{C}_{PCB}$ = 0.97 mg/kg

  $\bar{C}_{Hg}$ = 0.02 mg/kg

Determine Average Concentrations of Contaminants in Marine Organisms Based on 30-Day Exposures to Dredged Material In Laboratory Bioassays--

- Selected species used in bioaccumulation tests included English sole (Parophrys vetulus), clams (Macoma balthica), shrimp (Pandalus borealis), and polychaete worms (Neanthes arenaceodentata) (e.g., U.S. COE 1985).

- Determine average and range of concentrations (whole-body, wet-weight basis) for all species combined:

  [PCBs] = 0.242 ug/g (0.146 - 0.364 ug/g)
  [Mercury] = 0.31 ug/g (0.010 - 0.790 ug/g)

Characterize Exposed Populations

Humans--

- Potential exposure pathways included contact with dredged material deposited on the shore and consumption of contaminated marine organisms. Contact with dredged material can be ignored because results of the transport model showed that dredged material was not deposited on the shore (Figure 13 above).

- The potentially exposed population is the recreational public that harvests and consumes fish, crabs, and clams within the Elliott Bay area. Recreational fishing areas along the shore are shown in Figure 12 above. Information on fishing activity, catch-consumption patterns, and demographic variables for this potentially exposed population is given by McCallum (1985) and Landolt et al. (1985). Sensitive subgroups have not been characterized. Data limitations preclude precise estimation of seafood consumption rates.

- The recreational harvest in Elliott Bay is dominated by sablefish (62 percent), pollock (11 percent), Pacific cod (10 percent), and flatfish (7 percent), including English sole (McCallum 1985). For four urban areas, including Elliott Bay as characterized by Landolt et al. (1985), the dominant identifiable species in the catch were various salmon (28 percent), market squid (12 percent), Pacific hake (6.5 percent), and Pacific cod (6.5 percent). English sole accounted for only 0.6 percent of the total catch noted by Landolt et al. (1985). (Percent of catch by weight is shown in parentheses above).

- The proportion of the fisheries catch that comes in contact with the deepwater Fourmile Rock Disposal Site is unknown.

- Typical average seafood consumption rates (all species combined) may be on the order of 10-20 g/day.

- The range of average seafood consumption rates for exposure analysis is assumed to be:

  6.5 g/day = National (U.S.) average consumption of estuarine and freshwater fish, a standard assumption used in risk analysis (U.S. EPA 1980b).

  20 g/day = High estimate of average seafood consumption rate for Elliott Bay anglers.

Populations of Nonhuman Indicator Species--

- The amphipod (Rhepoxynius abronius) is a nonmigrant ubiquitously distributed species that inhabits deepwater and shallow-water sediments in Elliott Bay. Typical population densities

are on the order of 50 individuals/m^2, with a range of about 0-150 individuals/m^2. Because predisposal data on amphipod abundance within the disposal area are unavailable, abundance (N_{xr}) is assumed equal to 50/m^2 for all locations. The exposed population is located only within the disposal site (Figure 14).

- The geoduck (Panope generosa) is a nonmigrant, discretely distributed species. A major bed of geoducks is located just northeast of the deepwater disposal site (Figure 14). Population density estimates are unavailable. The exposed population is in a small area where the species distribution overlaps the outer area of the disposal site (Figure 14). [Note that this assumed distribution is hypothetical. The actual geoduck population does not overlap the designated Fourmile Rock Disposal Site.]

- English sole (Parophrys vetulus) is a small-scale, migrant species, with a ubiquitous population distribution. Although English sole are found in a variety of habitats, abundant local populations occur primarily in association with fine-grained muddy sediments. Because predisposal abundance data are unavailable, population density (N_{xr}) is assumed equal to 0.004/m^2 for all locations based on data from the literature on densities in Elliott Bay off Magnolia Bluff. The distribution of the hypothetical exposed population was determined as follows. The appropriate size of the home range of an individual was assumed equal to the cross-hatched area in Figure 14, based on a literature review. Because English sole exhibit seasonal migrations perpendicular to the shore, the example home range extends to deepwater areas (approximately bounded by the 600 ft depth contour in Figure 14). The cross-hatched area in the figure represents the home range of an individual that occasionally enters an infinitesimally small area of the dredged material disposal site. The contiguous home range of the next individual was displaced slightly downshore and did not overlap the disposal site. Similarly, an individual home range was placed just northwest of the border of the disposal site. All areas between the northwestern border of this individual home range and the southeastern border of the other individual home range were considered occupied by exposed individuals. Thus, the exposed population can be modeled as a series of individuals with overlapping home ranges. Individual home ranges may overlap completely, partially, or not at all depending on their relative locations.

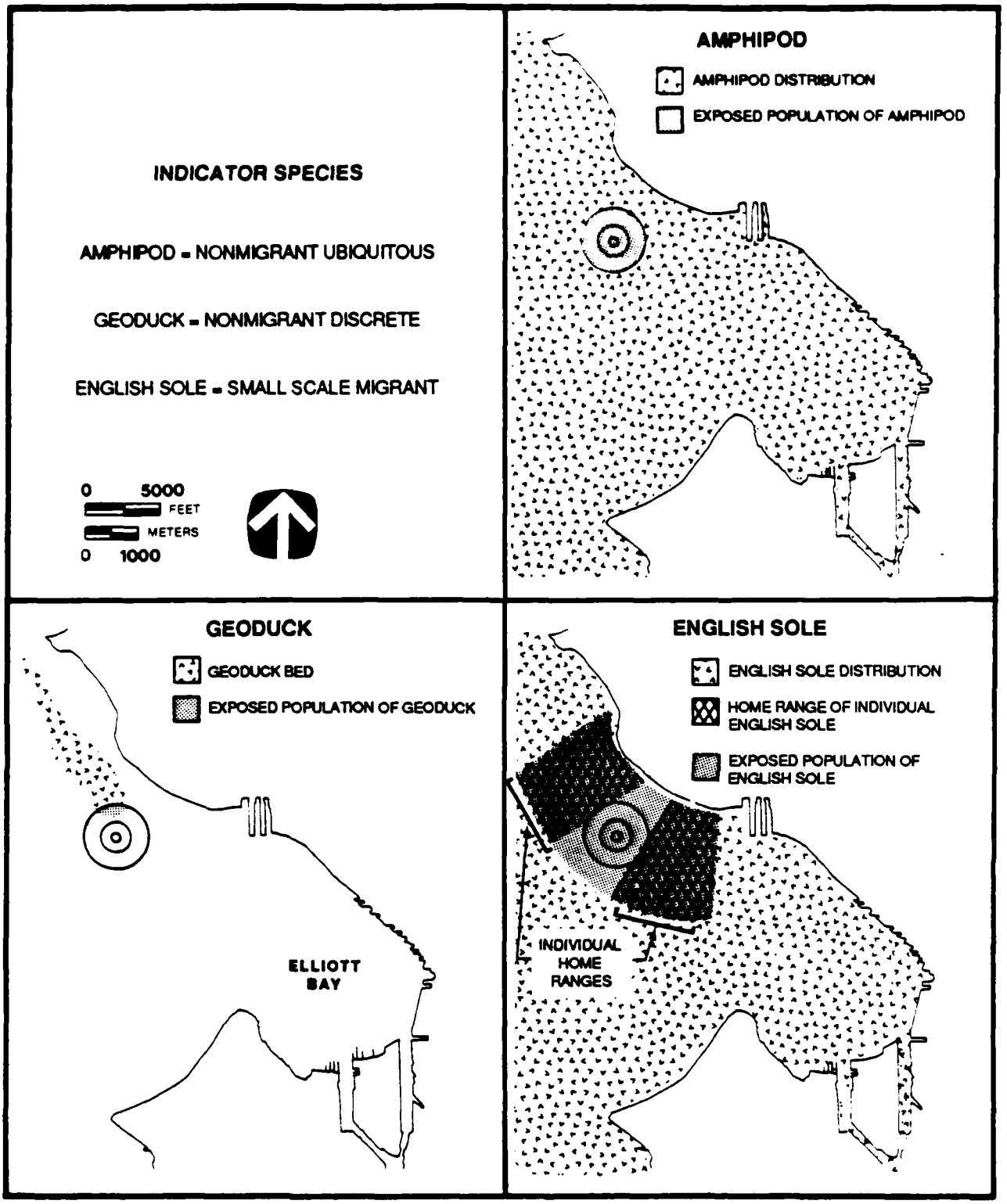


Figure 14. Hypothetical population distributions of potentially exposed indicator species.

Perform Integrated Exposure Analysis

Humans--

- Food ingestion is the only exposure route of concern.
- Assume the average concentrations of contaminants in whole-body samples of marine species above represents those in species and tissues potentially eaten by humans.
- Estimates of exposure variables (C_{ix}, I_{jk}) determined in previous steps are shown in Table 9. All absorption coefficients (X) are assumed equal to 1.0. Reference body weight (w) is assumed equal to 70 kg. The selected species (media) are English sole, Macoma clams, Pandalus shrimp, and polychaete worms. Concentrations of contaminants were taken from an example data set in U.S. COE (1985) (see above, Estimate Exposure Concentrations).
- Calculate range of exposure values (= ingested dose) for dredged material disposal case using Equation 7 and average, minimum, and maximum concentrations of PCBs and mercury in seafood, and 6.5 g/day and 20 g/day consumption rates (Table 9). Repeat calculations for reference-area case using contaminant concentrations in organisms exposed to reference sediment (Table 10).

  -- Disposal Site Exposure

  PCB intake = 1.4×10^{-5} to 1.0×10^{-4} $mg \cdot kg^{-1} \cdot day^{-1}$
  Mercury intake = 9.3×10^{-7} to 2.3×10^{-4} $mg \cdot kg^{-1} \cdot day^{-1}$

  -- Reference Area Exposure

  PCB intake = 3.7×10^{-7} to 2.9×10^{-6} $mg \cdot kg^{-1} \cdot day^{-1}$
  Mercury intake = 7.4×10^{-7} to 1.4×10^{-4} $mg \cdot kg^{-1} \cdot day^{-1}$

- Calculate excess exposure (equals average Disposal Site Exposure minus average Reference Area Exposure)

  Excess PCB intake (E_{PCB}) = 4.4×10^{-5} $mg \cdot kg^{-1} \cdot day^{-1}$
  Excess Mercury intake (E_{Hg}) = 2.9×10^{-5} $mg \cdot kg^{-1} \cdot day^{-1}$

Nonhuman Species--

Determine Exposure Probability by Area and by Species--

- Disposal site areas (see above, Figure 13):

  $A(1) = 0.30$ km^2

  $A(2) = 0.16$ km^2

TABLE 9. QUANTITATIVE RISK ASSESSMENT FOR CONSUMPTION OF SEAFOOD FROM A DISPOSAL SITE

	Exposure Determination							Risk Determination			
								Carcinogens		Noncarcinogens	
Substance	Concentration in Medium (mg/kg) [a]	Contact Rate (g/day) [b]	Total Daily Contact (mg/day)	Exposure Duration (years)	Absorption Coefficient (0-1.0) [c]	Body Weight (kg)	Exposure Value (mg/kg/d)	Potency Factor 1/(mg/kg/d)	Risk	ADI (mg/kg/d)	Hazard Index
PCBs	0.242	6.5	1.6E-03	70.0	1.0	70	2.2E-05	4.34	9.8E-05	N/A [d]	N/A
	0.146	6.5	9.5E-04	70.0	1.0	70	1.4E-05	4.34	5.9E-05	N/A	N/A
	0.364	6.5	2.4E-03	70.0	1.0	70	3.4E-05	4.34	1.5E-04	N/A	N/A
PCBs	0.242	20.0	4.8E-03	70.0	1.0	70	6.9E-05	4.34	3.0E-04	N/A	N/A
	0.146	20.0	2.9E-03	70.0	1.0	70	4.2E-05	4.34	1.8E-04	N/A	N/A
	0.364	20.0	7.3E-03	70.0	1.0	70	1.0E-04	4.34	4.5E-04	N/A	N/A
Hg	0.310	6.5	2.0E-03	70.0	1.0	70	2.9E-05	N/A	N/A	2.9E-04	1.0E-01
	0.010	6.5	6.5E-05	70.0	1.0	70	9.3E-07	N/A	N/A	2.9E-04	3.3E-03
	0.790	6.5	5.1E-03	70.0	1.0	70	7.3E-05	N/A	N/A	2.9E-04	2.6E-01
Hg	0.310	20.0	6.2E-03	70.0	1.0	70	8.9E-05	N/A	N/A	2.9E-04	3.1E-01
	0.010	20.0	2.0E-04	70.0	1.0	70	2.9E-06	N/A	N/A	2.9E-04	1.0E-02
	0.790	20.0	1.6E-02	70.0	1.0	70	2.3E-04	N/A	N/A	2.9E-04	7.9E-01

[a] Concentration of contaminant in exposure medium (mg/kg = ppm by mass).

[b] Amount of contaminated medium contacted or ingested per day, prior to accounting for absorption efficiency, etc.

[c] Ratio of g of contaminant absorbed per g of contaminant contacted.

[d] N/A = not applicable.

TABLE 10. QUANTITATIVE RISK ASSESSMENT FOR CONSUMPTION OF SEAFOOD FROM A REFERENCE AREA

	Exposure Determination							Risk Determination			
								Carcinogens		Noncarcinogens	
Substance	Concentration in Medium (mg/kg) a	Contact Rate (g/day) b	Total Daily Contact (mg/day)	Exposure Duration (years)	Absorption Coefficient (0-1.0) c	Body Weight (kg)	Exposure Value (mg/kg/d)	Potency Factor 1/(mg/kg/d)	Risk	ADI (mg/kg/d)	Hazard Index
PCBs	0.007	6.5	4.6E-05	70.0	1.0	70	6.5E-07	4.34	2.8E-06	N/A d	N/A
	0.004	6.5	2.6E-05	70.0	1.0	70	3.7E-07	4.34	1.6E-06	N/A	N/A
	0.010	6.5	6.5E 05	70.0	1.0	70	9.3E-07	4.34	4.0E-06	N/A	N/A
PCBs	0.007	20.0	1.4E-04	70.0	1.0	70	2.0E-06	4.34	8.7E-06	N/A	N/A
	0.004	20.0	8.0E-05	70.0	1.0	70	1.1E-06	4.34	5.0E-06	N/A	N/A
	0.010	20.0	2.0E-04	70.0	1.0	70	2.9E-06	4.34	1.2E-05	N/A	N/A
Hg	0.157	6.5	1.0E-03	70.0	1.0	70	1.5E-05	N/A	N/A	2.9E-04	5.1E-02
	0.008	6.5	5.2E-05	70.0	1.0	70	7.4E-07	N/A	N/A	2.9E-04	2.6E-03
	0.478	6.5	3.1E-03	70.0	1.0	70	4.4E-05	N/A	N/A	2.9E-04	1.6E-01
Hg	0.157	20.0	3.1E-03	70.0	1.0	70	4.5E-05	N/A	N/A	2.9E-04	1.6E-01
	0.008	20.0	1.6E-04	70.0	1.0	70	2.3E-06	N/A	N/A	2.9E-04	8.0E-03
	0.478	20.0	9.6E-03	70.0	1.0	70	1.4E-04	N/A	N/A	2.9E-04	4.8E-01

a Concentration of contaminant in exposure medium (mg/kg = ppm by mass).

b Amount of contaminated medium contacted or ingested per day, prior to accounting for absorption efficiency, etc.

c Ratio of g of contaminant absorbed per g of contaminant contacted.

d N/A = not applicable.

$A(3) = 1.30\ km^2$

$A(x) = 1.50\ km^2$

- Define probability of exposure (PE_{xr}) for nonmigrant species, amphipod and geoduck:

  PE_{xr} = 1.0 for all individuals located in areas within disposal site

  PE_{xr} = 0 for all individuals outside of disposal site

- Calculate exposure probability (PE_{xr}) for migrant English sole:

  From Figure 14:

  $O_{1r} = A(1) = 0.03\ km^2$

  $O_{2r} = A(2) = 0.16\ km^2$

  $O_{3r} = A(3) = 1.50\ km^2$

  $A_r = 11.2\ km^2$

  then, from Equations 4 and 5:

  $PE_{1r} = 0.003$

  $PE_{2r} = 0.014$

  $PE_{3r} = 0.134$

  $PE_r = 0.15$

Assume Exposure Duration--

- Exposure duration is equal to the length of sediment bioassay exposure (10 days).

RISK CHARACTERIZATION

Estimate Human Health Risk

Calculate Excess Carcinogenic Risk from the Exposure Dose and Carcinogenic Potency Factor for PCBs, Assuming Exposure Probability Equals 1.0 (Equations 6 and 7)--

- The range of plausible upper limits to excess lifetime risk of cancer for an individual exposed under the assumed conditions for the disposal case equals (see Table 9 above):

3×10^{-6} for minimum PCB concentration and 6.5 g/day consumption

1×10^{-5} for maximum PCB concentration and 20 g/day consumption.

Note that this range of estimates is derived from different exposure conditions. The plausible upper limit on carcinogenic potency was used to derive both the minimum and maximum risk estimates.

- For the example comparative risk analysis matrix below, the difference between average risk for the disposal case and average risk for the reference area is used as the best estimate of excess risk. Average excess risk calculated from data in Tables 9 and 10 equals:

$$PR_{PCB} = 2 \times 10^{-4}$$

Calculate an Index of Excess Noncarcinogenic Hazards from the Excess Exposure Dose and the RfD for Mercury, Assuming Exposure Probability Equals 1.0 (Equation 8)--

- The range of the hazard index for mercury intake by an individual exposed under the assumed conditions for the disposal case equals (see Table 9 above):

0.003 for minimum mercury concentration and 6.5 g/day consumption

0.8 for maximum mercury concentration and 20 g/day consumption.

- For the example comparative risk analysis matrix below, the difference between the hazard index for the disposal case and the average hazard index for the reference area case is used as the best estimate of excess hazard. Average excess hazard calculated from data in Tables 9 and 10 equals:

$$HI_{Hg} = 0.1$$

Estimate Ecological Risk

Calculate Site-Specific Probability of Excess Mortality (PM_x [illegible] of Encounter (PE_{xr}) and Probability of Excess Mortality [illegible] (C(x))] (Equation 12)--

- Calculate PB_{xr} [C(x)] using [illegible] probability of excess mortality [illegible] dredged material [C(x)] determined [illegible] Response Assessment above). [illegible] the average concentration [illegible] in area x, as determined [illegible] (Figure 13 above).

AD-A184 859 FRAMEWORK FOR COMPARATIVE RISK ANALYSIS OF DREDGED MATERIAL DISPOSAL OPTIONS(U) TETRA TECH INC BELLEVUE WA R A PASTOROK OCT 86 TC3090-02 DACW67-85-D-0029 2/2

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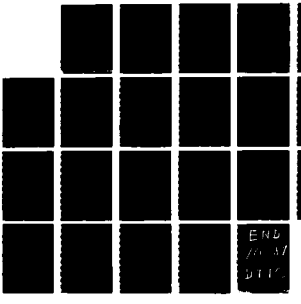

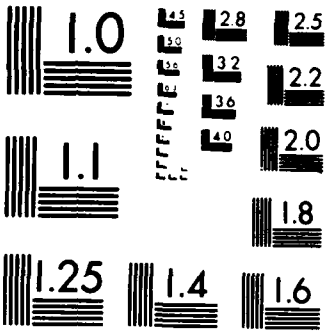

-- Amphipod: PB_{1r} (100%) = 0.75

PB_{2r} (10%) = 0.06

PB_{3r} (2%) = 0.01

- English sole: PB_{1r} (100%) = 0.50

PB_{2r} (10%) = 0.02

PB_{3r} (2%) = 0

- Calculate PM_{xr} using Equation 12:

-- Amphipod: PM_{1r} = 0.75

PM_{2r} = 0.06

PM_{3r} = 0.01

- English sole: PM_{1r} = 0.0015

PM_{2r} = 0.00028

PM_{3r} = 0

- Calculate average individual probability of excess mortality (PM_{xr} or PM_r) for entire disposal site (Equations 14 and 16):

-- Amphipod: PM_{xr} = 0.03

-- English Sole: PM_r = 0.002

- Note that physical impacts of dredged material disposal (e.g., burial) are ignored in this example.

Calculate Population Impact (I_r) Due to Dredged Material Disposal--

- Amphipod (Equation 17):

Given $N_{xr} = 50/m^2$ for all x locations

$A(1) = 0.03 \times 10^6\ m^2$

$A(2) = 0.16 \times 10^6\ m^2$

$A(3) = 1.30 \times 10^6\ m^2$

PM_{xr} as above

Then $I_r = 2 \times 10^6$ individuals.

- English sole (Equation 15):

  Given $N_{xr} = 0.004/m^2$ for all x locations

  $A_r = 11.2\ km^2$

  PM_{xr} as above

  Then $I_r = 80$ individuals.

- Geoduck:

  Because estimates of population abundance were not available, an index of population impact was calculated as the area of overlap between the population distribution and the disposal site divided by the area of population distribution

  -- Geoduck Index = $\frac{0.07\ km^2}{1.80\ km^2} = 0.04$

COMPARATIVE RISK ANALYSIS

Summarize Estimates of Risk and Population Impact

- Elements of the comparative analysis matrix are shown in Table 11. Note that only relative risk is evaluated in this example. Net cost-risk ratios are not considered. In practice, more exposure pathways and more variables would probably be considered in the matrix.

- Terms used in model equations for example calculations above are also shown in Table 11. Elevation Above Reference (EAR) values for sediment chemistry were calculated simply as the ratio of the concentration of the chemical in the dredged material to the concentration in reference-area sediment (data from Appendix E, Table E-1). Values of other variables for unconfined aquatic disposal were determined by previous calculations in the example analysis above. All risk index values for nearshore and upland disposal options would be determined by a similar approach. Hypothetical values are shown in this example (Table 11). Note that ecological and human health risk values for nearshore and upland disposal options may differ because of different site characteristics and exposure routes.

Create Comparative-Risk Scales

- In MATS, the range of each variable on the comparative-risk scale is defined as 0-1.0. The original range of the variable is termed the impact scale (Brown and Valenti 1983). For this example, the impact scale of each variable is defined as 0.0 to the maximum value of the variable within a row of Table 11.

TABLE 11. EXAMPLE COMPARATIVE RISK ANALYSIS MATRIX

Variable	Original Units	Risk Index Units	Model Term	Risk Index Values Aquatic Disposal	Nearshore Disposal	Upland Disposal
Sediment chemistry	mg/kg dry	unitless EAR	CS_i			
Bulk sediment						
PCBs				80	80	80
Mercury				5	5	5
Leachate						
PCBs				NA[a]	10	10
Mercury				NA	7	7
Human health risk						
Food						
PCBs	ug/g wet[b]	excess risk[a]	PR_{ijk}	$2x10^{-4}$	$2x10^{-4}$	$2x10^{-8}$
Mercury	ug/g wet[b]	excess hazard[a]	HI_i^{jk}	0.1	0.1	0.5
Groundwater						
PCBs	ug/L	excess risk	PR_{ijk}	NA	NA	$5x10^{-3}$
Mercury	ug/L	excess hazard	HI_i^{jk}	NA	NA	1.2
Ecological impact						
Plant species	% mortality[c]	No. killed	I_r	NA	0	200
Earthworm	% mortality[c]	No. killed	I_r	NA	0	$2x10^4$
Wildlife	overlap area[d]	overlap index[c]	PE_r	NA	0.1	0.2
Amphipod	% mortality[c]	No. killed	I_r	$2x10^6$	$2x10^2$	NA
English sole	% mortality[c]	No. killed	I_r	80	0	NA
Geoduck	overlap area[d]	overlap index[c]	PE_r	0.04	0	NA

[a] NA = Not applicable.

[b] PCB lifetime cancer risk and mercury hazard were calculated from contaminant concentrations (ug/g wet) in tissue of marine organisms exposed to dredged material in laboratory tests. Mercury hazard is expressed as excess exposure (mg/kg/day) as a proportion of the RfD. See text above for all calculations.

[c] Excess percent mortality in dredged material bioassay test relative to reference area sediment.

[d] Overlap index equals area of overlap between resource (e.g., wildlife habitat, geoduck bed) and disposal site divided by area of resource distribution within study area.

Create Comparative-Risk Function

- For simplicity in this example, each comparative-risk function is assumed to be a straight line with a slope of 1.0 and an intercept of 0.0. Thus, the value of the variable on the comparative risk scale is simply the value shown in Table 11 divided by the maximum value of the variable within the corresponding row of the table. This normalization procedure should be applied throughout the analysis. If one or more alternatives are dropped during the analysis, the original impact scales and corresponding comparative-risk scales should be maintained. That is, the data should not be re-normalized to the new maximum values of the variables.

Assign Weights to Factors

- Weights given to each variable in MATS sum to 1.0. Variable weights are assumed to depend on disposal option for this example. The application of varying weights for a single factor across plan alternatives is not a common practice among users of MATS. However, the use of varying weights across disposal alternatives is justified in light of the different numbers and kinds of factors across alternatives. Because of large differences in ecological systems and exposure pathways among disposal alternatives, evaluation of a single set of factors with constant weights is inappropriate. If the weights were held constant across alternatives, many factors that are unique to a subset of disposal options would have to be ignored. Otherwise, the rank score for alternatives with less data would be biased toward low values. In the example shown in Table 12, individual factors are weighted less heavily as more factors are included in the evaluation of a given option.

- In practice, several different sets of weights should be used to aid in identifying differences among alternatives. Different sets of factor weights might reflect different socio-political perspectives. Sensitivity analysis using different assumptions for factor weights is not shown in this example, but would be performed by applying all subsequent steps using each set of weights. Brown and Valenti (1983) provide examples of sensitivity analysis using MATS.

- For any cell in the matrix Table 11 that shows NA (not applicable), the weight of that factor for that disposal option is 0.0.

- Weights shown in Table 12 are assumed hypothetical values. Actual weights would be determined by administrative policy, with public input. One mechanism for derivation of factor weights would be a workshop or Delphi survey of administrators, agency technical personnel, university scientists, and public representatives.

TABLE 12. WEIGHTS APPLIED TO VARIABLES USED IN EXAMPLE OF MULTI-ATTRIBUTE TRADEOFF SYSTEM MODEL

Variable	Weight of Variable		
	Unconfined Aquatic Disposal	Nearshore Disposal	Upland Disposal
Sediment chemistry			
Bulk sediment			
PCBs	0.15	0.05	0.05
Mercury	0.15	0.05	0.05
Leachate			
PCBs	0	0.1	0.1
Mercury	0	0.1	0.1
Human health risk			
Food			
PCBs	0.2	0.2	0.05
Mercury	0.2	0.2	0.05
Groundwater			
PCBs	0	0	0.15
Mercury	0	0	0.15
Ecological impact			
Plant species	0	0.05	0.1
Earthworm	0	0.025	0.05
Wildlife	0	0.075	0.15
Amphipod	0.05	0.025	0
English sole	0.1	0.05	0
Geoduck	0.15	0.075	0

Calculate Weighted Risk Variables

- Weighted comparative-risk variables were calculated for each entry in the matrix. For example, for wildlife impact under Nearshore Disposal:

$$\text{Weighted Risk Index} = \frac{0.1}{0.2} \quad (0.075) = 0.0375$$

Summarize Recommendations

- The final step in the analysis is to calculate the total sum of weighted risk indices within a column for each disposal option. The maximum possible value of this net rank score is 1.0. In the sensitivity analysis, this step would be performed for each set of factor weights.

- The example net rank scores are shown below:

Unconfined Aquatic Disposal:	0.84
Nearshore Disposal:	0.58
Upland Disposal:	0.95

Based on relative risk analysis, the nearshore site is the preferred disposal option, followed by the unconfined aquatic disposal site and the upland disposal site. In practice, use of several sets of factor weights in a sensitivity analysis would result in a range of alternative net rank scores. The similarity of the ranking of disposal options for each alternative set of factor weights would indicate the robustness of the solution. When the solution to the model is not robust, professional judgment and cost considerations may weigh more heavily in final decision-making.

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APPENDIX A

GUIDANCE FOR INTERPRETATION OF BIOACCUMULATION DATA

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GUIDANCE FOR INTERPRETATION OF BIOACCUMULATION DATA

Uptake of chemical contaminants into aquatic and terrestrial food webs may ultimately result in human exposure to potentially toxic agents. Exposure of nonhuman organisms to dredged material deposited in the environment is therefore one mechanism accounting for potential human health risks from dredged material disposal. Before disposal, the current testing strategies for sediments dredged from Puget Sound include measurement of the uptake of contaminants by organisms (e.g., clams, earthworms) exposed to the sediments in the laboratory (U.S. COE 1985). The results of bioaccumulation tests on dredged material usually consist of tables of "steady-state" or final concentrations of contaminants in tissues of organisms exposed for a controlled period of time. The exposed organisms may or may not be potential food items for humans. The present approach to evaluation of the data, however, treats the organisms as if they were to be consumed by humans. This conservative approach is warranted because of the potential for food chain transfer of most contaminants of concern, and possibly biomagnification of some organochlorine compounds (e.g., DDT, PCB).

The objectives of the following analysis are to:

- Describe the limitations of the present approach to evaluation of bioaccumulation data relative to human health concerns
- Develop guidance for evaluation of bioaccumulation data based on a generic application of human health risk assessment models
- Determine conservative (i.e., protective) guidelines for maximum allowable concentrations of selected chemical contaminants in dredged material.

In the following analysis, risk assessment models are used to develop reference guidelines for concentrations of toxic contaminants in marine organism tissues. The tissue contamination guidelines for carcinogens are derived from assumed lifetime cancer risk levels in hypothetically exposed humans. A similar approach is used to derive guidelines for tissue concentrations of noncarcinogens except that a Reference Dose (RfD in units $mg \cdot kg^{-1} \cdot day^{-1}$) value for hypothetically exposed humans is used in place of an assumed risk. If the assumed risk level or the RfD is regarded as a maximum acceptable value, the derived tissue contamination guidelines may be viewed as maximum acceptable concentrations of contaminants. That is, when the concentrations of a given contaminant is below its guideline, the potential human health effects associated with assumed seafood consumption rates are considered tolerable.

The tissue contamination guidelines derived below would be compared with body burden data from laboratory exposures of organisms to dredged material. This evaluation technique would be appropriate for use as a screening tool when site-specific risk assessment methods are not required or data are not available. Note that this generic approach to evaluation of human health risks considers only hypothetical exposure scenarios. Actual human exposures will depend on site-specific conditions and dredged material composition. Tetra Tech (1986b) provides further guidance for assessing human health risks from consumption of chemically contaminated seafood.

LIMITATIONS OF U.S. FDA ACTION LEVELS FOR EVALUATION OF BIOACCUMULATION DATA

One approach to the evaluation of bioaccumulation data is to compare tissue concentrations of contaminants in edible marine organisms with regulatory limits used to protect consumers. At present, the evaluation of dredged-material bioaccumulation data relative to human health concerns generally consists of comparison of tissue concentrations of selected toxic chemicals to action levels or tolerances established by the U.S. Food and Drug Administration (U.S. FDA 1982, 1984; U.S. COE 1985). However, this approach is severely limited for the following reasons:

- U.S. FDA limits are available for only a few chemicals (e.g., mercury and 13 organic compounds)
- U.S. FDA has not established regulatory limits for some of the most potent suspected human carcinogens [e.g., 2,3,7,8-tetrachlorodibenzodioxin, benzo(a)pyrene]
- Action levels and tolerances were intended to be used only for regulation of interstate commerce of food products
- In setting regulatory limits, U.S. FDA considers economic impacts of food regulation as well as potential human health risk. When using U.S. FDA limits to interpret bioaccumulation data, economic policies of U.S. FDA are implicitly adopted by the investigator. Thus, risk management issues are not clearly separated from risk assessments.

Countries other than the U.S. have also developed regulatory limits on toxic chemicals in food products (Nauen 1983). As a whole, regulatory limits on contaminants in fishery products are available for 9 metals and 17 organic compounds of potential concern in dredged material assessments. Nevertheless, use of regulatory limits established by other countries would suffer from many of the limitations listed above for U.S. FDA values. Moreover, a concise review of the basis for each of these limits is not available.

APPROACH TO DERIVATION OF TISSUE CONTAMINATION GUIDELINES

The following sections describe the procedures used to calculate tissue contamination guidelines. Associated uncertainties and assumptions are also discussed. Quantitative uncertainty analysis is beyond the scope of work. U.S. EPA (1980b, 1984a, 1985a) provides detailed descriptions of health risk assessment models and approaches outlined below.

Carcinogenic Risk Model

Excess lifetime risk (R_i) of cancer due to exposure of an individual to chemical i is calculated as the product of a carcinogenic potency factor (B_i, in $kg \cdot day \cdot mg^{-1}$) and an exposure estimate (E_i, in $mg \cdot kg^{-1} \cdot day^{-1}$, or mg of chemical i per kg of consumer's body weight per day):

$$R_i = B_i E_i \quad (1)$$

and:

$$E_i = \frac{C_i I}{W} \quad (2)$$

where:

C_i = Average concentration of carcinogenic chemical i in edible tissue of a seafood organism (mg/kg)

I = Average seafood ingestion rate per human (kg/day)

W = Reference human weight (kg).

W is assumed to be 70 kg for the "reference man" (U.S. EPA 1980b). An average human lifetime is assumed to be 70 yr. Carcinogenic potency factors were obtained from U.S. EPA (1985a) (see Appendix C, Table C-1).

Given a reference-risk level (R_i^*) and an average consumption rate (I^*), the corresponding tissue-concentration guideline (C_i^*) of a single chemical can be calculated by combining Equations 1 and 2 and solving for C_i^* as follows:

$$C_i^* = \frac{R_i^* W}{B_i I^*} \quad (3)$$

That is, the tissue concentration guideline for chemical i is equal to the reference-risk level times the reference body weight (70 kg) divided by the product of carcinogenic potency of chemical i and the assumed average ingestion rate. Note that values for I^* and R^* are established by policy decisions. The assumed values for consumption rate (I^*) and risk (R_i^*) are discussed in the following sections.

Reference-Risk Value

Reference-risk levels of 10^{-4}, 10^{-5}, and 10^{-6} (lifetime cancer risk of 1 in 10,000, 1 in 100,000, and 1 in 1,000,000) were used to calculate tissue contamination criteria. By comparison, U.S. EPA (1980b) has used lifetime risk levels of 10^{-5} to 10^{-7} to develop water quality criteria. In general, U.S. EPA has made decisions to allow levels of carcinogens in the environment where the estimates of individual lifetime risk have been within the range of 10^{-4} to 10^{-8} (Thomas 1984). Note that U.S. EPA has avoided defining a single "acceptable risk" in deriving water quality criteria for carcinogens, because methods are not available for establishing the presence of a threshold for carcinogenesis. The reference-risk level and corresponding tissue contamination guidelines presented in this report should not necessarily be interpreted as "safe" levels, but rather as reference values.

Everyday exposure to common carcinogens may lead to average lifetime cancer risks as high as 7×10^{-4} (Table A-1). Cigarette smokers experience higher risks; e.g., on the order of 8×10^{-2} for average smokers. Risks on the order of 7×10^{-5} per lifetime (10^{-6}/yr) are commonly considered acceptable, while higher risks are clearly of concern to environmental regulators (Pochin 1975; Crouch et al. 1983). In setting standards for benzene exposure, Justice Stewart of the U.S. Supreme Court argued that lifetime risks of 10^{-3} were clearly "unacceptable," whereas those of 10^{-9} were clearly "acceptable" (Connor 1983).

Seafood Ingestion Rate

The average ingestion rates used to calculate tissue contamination guidelines were 6.5 g/day, 20 g/day (which equals approximately 0.33 lb/wk, or about one average serving per week), and 165 g/day. Estimates of average seafood consumption rate were obtained from the literature. The 6.5 g/day estimate is the value used by U.S. EPA (1980b) to derive water quality criteria. It represents the average per capita consumption of commercial fish and shellfish from estuarine and fresh waters in the U.S. based on data from National Marine Fisheries Service (1976). The U.S. Department of Agriculture (Johnson, E., 14 August 1984, personal communication) estimates that the average U.S. per capita consumption of commercial and recreational "seafood" from estuarine, marine, and freshwaters is about 20 g/d[illegible] (also see National Marine Fisheries Service 1984). The 165 g/day estimate represents the average rate of consumption of commercial seafood by a small portion (about 0.1 percent) of the U.S. population (Finch 1973). Note that inclusion of freshwater species in the original data sets does not bias the analysis substantially. For example, freshwater species account for 1.7 g/day consumption, or 9 percent of the total 18.7 g/day consumption of fish and shellfish harvested commercially from estuarine, marine, and fresh waters (National Marine Fisheries Service 1976).

TABLE A-1. EXAMPLES OF CANCER RISKS FROM COMMON CARCINOGENS

	Average Lifetime Risk[a]	Average Annual Risk	Uncertainty
Diet soda (saccharin) - 12.5 oz/day	7×10^{-4}	1×10^{-5}	Factor of 10
Average saccharin consumption	1×10^{-4}	2×10^{-6}	
Peanut butter (aflatoxins) - 4 tbsp/day	6×10^{-4}	8×10^{-6}	
Milk (aflatoxins) - 1 pt/day	1×10^{-4}	2×10^{-6}	
Miami/New Orleans drinking water - 2 L/day	7×10^{-5}	1×10^{-6}	
Charcoal broiled steak (PAH) - 0.5 lb/wk	2×10^{-5}	3×10^{-7}	
Average smoker (PAH)[b]	8×10^{-2}	1.2×10^{-3}	Factor of 3
Person sharing room with smoker	7×10^{-4}	1×10^{-5}	Factor of 10

[a] Average lifetime risks were calculated from average annual risks during preparation of this report assuming a lifetime of 70 yr.

[b] Risk estimate based on human data.

Reference: Crouch and Wilson (1984).

Noncarcinogens

By substituting the RfD for a noncarcinogenic chemical in Equation 2 and specifying an average ingestion rate (I*), the tissue contamination guideline may be calculated as follows:

$$C_i^* = \frac{RfD(W)}{I^*} \quad (4)$$

As before, average body weight (W) is assumed to be 70 kg and the average ingestion rate (I*) is 6.5 g/day, 20 g/day, or 165 g/day.

Reference Dose (RfD) values were obtained from U.S. EPA (1980b, 1986) and the Environmental Criteria and Assessment Office, U.S. EPA, Cincinnati, Ohio (see Appendix C, Table C-2). Although RfD values were published as part of water quality criteria development (U.S. EPA 1980b), some of these values are now being revised. Values used in this assessment are the current published values, but they are subject to revision. Note that a tissue concentration guideline was not calculated for fluoranthene. The RfD for fluoranthene is based on the dermal route of exposure and may not be applicable to exposure by ingestion.

SUMMARY OF ASSUMPTIONS

Assumptions and estimated values for model variables used in this analysis are summarized in Table A-2. Note that the methods and assumptions chosen for this analysis are conservative (i.e., protective of human health). For example, U.S. EPA (1980b, 1984c, 1985a) uses a conservative approach to derive carcinogenic potency factors and RfD values, so the final results are protective of human health. Detailed discussion of assumptions made in estimating potency factors and RfD values can be found in U.S. EPA (1980b, 1985a) and Dourson and Stara (1983).

UNCERTAINTIES

Uncertainties are inherent in all risk assessments (e.g., Crouch et a'. 1983; U.S. EPA 1984a,b, 1985a,b). Uncertainties in the present analysis arise from the following factors:

1. Uncertainties in estimating carcinogenic potency factors or RfDs, resulting from

   - Uncertainties in extrapolating from toxicologic data obtained from laboratory animals to humans

   - Uncertainties in high- to low-dose extrapolation of bioassay test results, which arise from practical limitations of laboratory experiments.

2. Uncertainties in the selection of 6.5 g/day, 20 g/day, and 165 g/day as average consumption rates.

TABLE A-2. SUMMARY OF ASSUMPTIONS AND NUMERICAL ESTIMATES USED IN DERIVATION OF TISSUE CONTAMINATION GUIDELINES

Parameter	Assumptions/Estimates	Reference
Exposure Assessment:		
Contaminant concentrations in tissues of indicator species	No effect of cooking	Worst case for parent compounds. Net effect on risk is uncertain.
Average consumption rate	6.5 g/day 20 g/day 165 g/day	Low, moderate, and high values specified by regulatory policy (see text)
Gastrointestinal absorption coefficient	1.0 Assumes contaminant absorption efficiencies of humans and laboratory bioassay animals are equal	U.S. EPA 1984[a]
Exposure duration	70 yr	U.S. EPA-CAG[a] U.S. EPA 1980b
Human body weight	70 kg (= avg. adult male)	U.S. EPA-CAG
Risk Characterization:		
Carcinogenic risk model	Linearized Multistage Model (linear, no-threshold model). At risks less than 10^{-2}: Risk = Exposure x Potency.	U.S. EPA 1980b
Carcinogenic potency	Potency factors are based on low-dose extrapolation from animal bioassay data. Upper bound of 95 percent confidence interval on potency is used.	U.S. EPA 1980b, 1985a
Acceptable Daily Intakes (ADIs)	ADIs for noncarcinogens are current U.S. EPA values.	U.S. EPA 1980b, 1986; U.S. EPA Environmental Criteria and Assessment Office

[a] U.S. Environmental Protection Agency Carcinogen Assessment Group.

3. The efficiency of assimilation (or absorption) of contaminants by the human gastrointestinal system is unknown. It is assumed to be 100 percent in this study.

4. Uncertainties associated with variation of exposure factors among individuals, such as

   - Variation in seafood species composition of the diet among individuals

   - Variation in seafood preparation methods and uncertainties associated with changes in chemical concentrations due to cooking.

Variance in estimates of carcinogenic potency or RfDs (#1 above) account for one major uncertainty component in this study. Chemical potencies are estimated only on an order-of-magnitude basis, whereas analytical chemistry of tissues is relatively precise (on the order of $\pm$20 percent).

Because of data limitations, variance of the calculated tissue contamination guidelines can not be estimated precisely. However, uncertainty analysis conducted by previous researchers illustrates the variability of risk estimates and potency factors. For example, the coefficient of variation for the mean value of potency generally ranged from 2 to 105 percent for each drinking water contaminant studied by Crouch et al. (1983). This uncertainty arises mainly from error associated with experimental bioassay data. Among species, the potency of a given chemical may vary only slightly or up to approximately 1,000-fold, depending on the chemical in question (Clayson et al. 1983). Thus, the uncertainty associated with extrapolating estimates of potency from laboratory animals to humans may be much greater than the uncertainty associated with animal bioassay techniques. By comparison, the range of potencies among carcinogens covers seven to nine orders of magnitude (Clayson et al. 1983; U.S. EPA 1985a).

TISSUE CONTAMINATION GUIDELINES AND THEIR USE

Tissue contamination guidelines based on the models presented earlier are presented in this section. Uses of the guidelines are discussed briefly, including evaluation of chemical mixtures.

Guidelines for Carcinogens

Tissue contamination guidelines are shown in Tables A-3 to A-5 for selected upper-limit cancer risks (10^{-4}, 10^{-5}, and 10^{-6}) and seafood consumption rates 6.5, 20, and 165 g/day). At an assumed upper-limit risk level of 10^{-5}, the target tissue concentrations are less than 1 ppm for most of the carcinogenic priority pollutants. That is, if the concentration of the given chemical in seafood is greater than 1 ppm (on a wet-weight basis), the plausible-upper-limit-risk of cancer during a 70-yr lifetime is expected to be greater than 10^{-5}.

Note that guidelines are not available for some priority pollutants of concern in evaluation of dredged material [e.g., polynuclear aromatic hydrocarbons (PAH) other than benzo(a)pyrene]. For some of these substances, however, the toxicologic potency may be assumed to be similar to that determined for a related compounds. For example, the carcinogenic potency factor and tissue contamination guideline for benzo(a)pyrene may be used as interim values for other carcinogenic PAH. Such policy decisions should be made on a case-by-case basis, and generally require a Regional Authority Decision.

The guidelines for arsenic in Tables A-3 to A-5 apply only to inorganic arsenic, the form suspected of being carcinogenic in humans. Because most tissues will probably be analyzed for total arsenic only, it will be necessary to estimate the contribution of the inorganic form of arsenic to the total arsenic concentration. For example, Crecelius and Apts (1985) determined that inorganic forms accounted for an average of about 0.12 percent of total arsenic in a variety of fish and shellfish species from Commencement Bay and Discovery Bay. Based on this estimate, the guideline for total arsenic can be calculated from the inorganic arsenic guidelines in Tables A-3 to A-5 as follows:

$$\text{Total arsenic guideline} = \frac{\text{Inorganic Arsenic Guideline}}{0.0012}$$

For example, the total arsenic guideline at an upper-limit risk of 10^{-5} and a seafood consumption rate of 20 g/day (1 meal/wk) equals 1.9 ppm, or about 2 ppm.

Guidelines Based on Reference Dose (RfD) Values

Tissue contamination guidelines for priority pollutants with RfD values are shown in Table A-6 at selected seafood consumption rates (6.5, 20, and 165 g/day). At a seafood consumption rate of 20 g/day, the guidelines range from 0.8 ppm for silver to 40,000 ppm for diethyl and dimethyl phthalates.

TABLE A-3. TARGET CONCENTRATIONS (C_i*) OF CARCINOGENS AT SELECTED RATES OF SEAFOOD CONSUMPTION AND LIFETIME RISK OF 10^{-4}

PP#	Pollutant	Target Concentration (ppm)[a]		
		6.5 g/day	20 g/day	165 g/day
129	TCDD (dioxin)	0.000007000	0.000002000	0.000000300
5	benzidine	0.005000000	0.001000000	0.000200000
90	dieldrin	0.040000000	0.010000000	0.001000000
61	N-nitrosodimethylamine	0.040000000	0.010000000	0.002000000
115	arsenic	0.070000000	0.020000000	0.003000000
73	benzo(a)pyrene[b]	0.090000000	0.030000000	0.004000000
89	aldrin	0.090000000	0.030000000	0.004000000
102	alpha-HCH	0.100000000	0.030000000	0.004000000
106-112	PCBs	0.200000000	0.080000000	0.010000000
100	heptachlor	0.300000000	0.100000000	0.010000000
103	beta-HCH	0.600000000	0.200000000	0.020000000
28	3,3'-dichlorobenzidine	0.600000000	0.200000000	0.030000000
9	hexachlorobenzene	0.600000000	0.200000000	0.030000000
91	chlordane	0.700000000	0.200000000	0.030000000
105	gamma-HCH	0.800000000	0.300000000	0.030000000
29	1,1-dichloroethene	0.900000000	0.300000000	0.040000000
18	bis(2-chloroethyl)ether	0.900000000	0.300000000	0.040000000
113	toxaphene	1.000000000	0.300000000	0.040000000
37	1,2-diphenylhydrazine	1.000000000	0.500000000	0.060000000
92-94	4,4'-DDT, DDD, DDE	3.000000000	1.000000000	0.100000000
35	2,4-dinitrotoluene	3.000000000	1.000000000	0.100000000
3	acrylonitrile	4.000000000	1.000000000	0.200000000
15	1,1,2,2-tetrachloroethane	5.000000000	2.000000000	0.200000000
6	tetrachloromethane	8.000000000	3.000000000	0.300000000
10	1,2-dichloroethane	10.000000000	4.000000000	0.500000000
52	hexachlorobutadiene	10.000000000	5.000000000	0.500000000
23	chloroform	20.000000000	5.000000000	0.600000000
14	1,1,2-trichloroethane	20.000000000	6.000000000	0.700000000
85	tetrachloroethene	20.000000000	7.000000000	0.800000000
4	benzene	40.000000000	10.000000000	1.000000000
21	2,4,6-trichlorophenol	50.000000000	20.000000000	2.000000000
88	vinyl chloride	60.000000000	20.000000000	2.000000000
12	hexachloroethane	80.000000000	20.000000000	3.000000000
87	trichloroethene	100.000000000	30.000000000	4.000000000
62	N-nitrosodiphenylamine	200.000000000	70.000000000	9.000000000
44	dichloromethane	2000.000000000	600.000000000	70.000000000

[a] Target concentrations were calculated using Equations 1 and 2 in text.

[b] Values for benzo(a)pyrene apply to other carcinogenic polycyclic aromatic hydrocarbons: benzo(a)anthracene, benzo(b)fluoranthene, chrysene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

TABLE A-4. TARGET CONCENTRATIONS (C_i*) OF CARCINOGENS AT SELECTED RATES OF SEAFOOD CONSUMPTION AND LIFETIME RISK OF 10^{-5}

PP#	Pollutant	Target Concentration (ppm)[a]		
		6.5 g/day	20 g/day	165 g/day
129	TCDD (dioxin)	0.000000700	0.000000200	0.000000030
5	benzidine	0.000500000	0.000100000	0.000020000
90	dieldrin	0.004000000	0.001000000	0.000100000
61	N-nitrosodimethylamine	0.004000000	0.001000000	0.000200000
115	arsenic	0.007000000	0.002000000	0.000300000
73	benzo(a)pyrene[b]	0.009000000	0.003000000	0.000400000
89	aldrin	0.009000000	0.003000000	0.000400000
102	alpha-HCH	0.010000000	0.003000000	0.000400000
106-112	PCBs	0.020000000	0.008000000	0.001000000
100	heptachlor	0.030000000	0.010000000	0.001000000
103	beta-HCH	0.060000000	0.020000000	0.002000000
28	3,3'-dichlorobenzidine	0.060000000	0.020000000	0.003000000
9	hexachlorobenzene	0.060000000	0.020000000	0.003000000
91	chlordane	0.070000000	0.020000000	0.003000000
105	gamma-HCH	0.080000000	0.030000000	0.003000000
29	1,1-dichloroethene	0.090000000	0.030000000	0.004000000
18	bis(2-chloroethyl)ether	0.090000000	0.030000000	0.004000000
113	toxaphene	0.100000000	0.030000000	0.004000000
37	1,2-diphenylhydrazine	0.100000000	0.050000000	0.006000000
92-94	4,4'-DDT, DDD, DDE	0.300000000	0.100000000	0.010000000
35	2,4-dinitrotoluene	0.300000000	0.100000000	0.010000000
3	acrylonitrile	0.400000000	0.100000000	0.020000000
15	1,1,2,2-tetrachloroethane	0.500000000	0.200000000	0.020000000
6	tetrachloromethane	0.800000000	0.300000000	0.030000000
10	1,2-dichloroethane	1.000000000	0.400000000	0.050000000
52	hexachlorobutadiene	1.000000000	0.500000000	0.050000000
23	chloroform	2.000000000	0.500000000	0.060000000
14	1,1,2-trichloroethane	2.000000000	0.600000000	0.070000000
85	tetrachloroethene	2.000000000	0.700000000	0.080000000
4	benzene	4.000000000	1.000000000	0.100000000
21	2,4,6-trichlorophenol	5.000000000	2.000000000	0.200000000
88	vinyl chloride	6.000000000	2.000000000	0.200000000
12	hexachloroethane	8.000000000	2.000000000	0.300000000
87	trichloroethene	10.000000000	3.000000000	0.400000000
62	N-nitrosodiphenylamine	20.000000000	7.000000000	0.900000000
44	dichloromethane	200.000000000	60.000000000	7.000000000

[a] Target concentrations were calculated using Equations 1 and 2 in text.

[b] Values for benzo(a)pyrene apply to other carcinogenic polycyclic aromatic hydrocarbons: benzo(a)anthracene, benzo(b)fluoranthene, chrysene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

TABLE A-5. TARGET CONCENTRATIONS (C_i*) OF CARCINOGENS AT SELECTED RATES OF SEAFOOD CONSUMPTION AND LIFETIME RISK OF 10^{-6}

PP#	Pollutant	Target Concentration (ppm)[a]		
		6.5 g/day	20 g/day	165 g/day
129	TCDD (dioxin)	0.000000070	0.000000020	0.000000003
5	benzidine	0.000050000	0.000010000	0.000002000
90	dieldrin	0.000400000	0.000100000	0.000010000
61	N-nitrosodimethylamine	0.000400000	0.000100000	0.000020000
115	arsenic	0.000700000	0.000200000	0.000030000
73	benzo(a)pyrene[b]	0.000900000	0.000300000	0.000040000
89	aldrin	0.000900000	0.000300000	0.000040000
102	alpha-HCH	0.001000000	0.000300000	0.000040000
106-112	PCBs	0.002000000	0.000800000	0.000100000
100	heptachlor	0.003000000	0.001000000	0.000100000
103	beta-HCH	0.006000000	0.002000000	0.000200000
28	3,3'-dichlorobenzidine	0.006000000	0.002000000	0.000300000
9	hexachlorobenzene	0.006000000	0.002000000	0.000300000
91	chlordane	0.007000000	0.002000000	0.000300000
105	gamma-HCH	0.008000000	0.003000000	0.000300000
29	1,1-dichloroethene	0.009000000	0.003000000	0.000400000
18	bis(2-chloroethyl)ether	0.009000000	0.003000000	0.000400000
113	toxaphene	0.010000000	0.003000000	0.000400000
37	1,2-diphenylhydrazine	0.010000000	0.005000000	0.000600000
92-94	4,4'-DDT, DDD, DDE	0.030000000	0.010000000	0.001000000
35	2,4-dinitrotoluene	0.030000000	0.010000000	0.001000000
3	acrylonitrile	0.040000000	0.010000000	0.002000000
15	1,1,2,2-tetrachloroethane	0.050000000	0.020000000	0.002000000
6	tetrachloromethane	0.080000000	0.030000000	0.003000000
10	1,2-dichloroethane	0.100000000	0.040000000	0.005000000
52	hexachlorobutadiene	0.100000000	0.050000000	0.005000000
23	chloroform	0.200000000	0.050000000	0.006000000
14	1,1,2-trichloroethane	0.200000000	0.060000000	0.007000000
85	tetrachloroethene	0.200000000	0.070000000	0.008000000
4	benzene	0.400000000	0.100000000	0.010000000
21	2,4,6-trichlorophenol	0.500000000	0.200000000	0.020000000
88	vinyl chloride	0.600000000	0.200000000	0.020000000
12	hexachloroethane	0.800000000	0.200000000	0.030000000
87	trichloroethene	1.000000000	0.300000000	0.040000000
62	N-nitrosodiphenylamine	2.000000000	0.700000000	0.090000000
44	dichloromethane	20.000000000	6.000000000	0.700000000

[a] Target concentrations were calculated using Equations 1 and 2 in text.

[b] Values for benzo(a)pyrene apply to other carcinogenic polycyclic aromatic hydrocarbons: benzo(a)anthracene, benzo(b)fluoranthene, chrysene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

TABLE A-6. TARGET CONCENTRATIONS (C_i*) OF PRIORITY POLLUTANTS BASED ON ADI VALUES AT SELECTED RATES OF SEAFOOD CONSUMPTION

PP#	Pollutant	Target Concentration (ppm)[a] 6.5 g/day	20 g/day	165 g/day
126	silver	2	0.8	0.10
60	4,6-dinitro-o-cresol	4	1.0	0.20
127	thallium	4	1.0	0.20
56	nitrobenzene	5	2.0	0.20
42	bis(2-chloroisopropyl)ether	10	4.0	0.40
98	endrin	10	4.0	0.40
123	mercury	20	7.0	0.80
59	2,4-dinitrophenol	20	7.0	0.80
33	1,3-dichloropropene	30	9.0	1.00
119	chromium VI	30	9.0	1.00
95	alpha-endosulfan	40	10.0	2.00
96	beta-endosulfan	40	10.0	2.00
97	endosulfan sulfate	40	10.0	2.00
114	antimony	40	10.0	2.00
53	hexachlorocyclopentadiene	60	20.0	3.00
125	selenium	100	40.0	4.00
25	1,2-dichlorobenzene	100	50.0	6.00
26	1,3-dichlorobenzene	100	50.0	6.00
27	1,4-dichlorobenzene	100	50.0	6.00
7	chlorobenzene	200	50.0	6.00
2	acrolein	200	60.0	7.00
46	bromomethane	200	80.0	9.00
124	nickel	200	80.0	9.00
121	cyanide	200	70.0	8.00
64	pentachlorophenol	300	100.0	10.00
44	dichloromethane	600	200.0	30.00
38	ethylbenzene	1000	400.0	40.00
31	2,4-dichlorophenol	1000	400.0	40.00
65	phenol	1000	400.0	40.00
54	isophorone	2000	500.0	60.00
86	toluene	5000	1000.0	200.00
11	1,1,1-trichloroethane	6000	2000.0	200.00
45	chloromethane	6000	2000.0	200.00
66	bis(2-ethylhexyl)phthalate	6000	2000.0	300.00
68	di-n-butyl phthalate	10000	4000.0	500.00
119	chromium III	20000	6000.0	800.00
71	dimethyl phthalate	100000	40000.0	4000.00
70	diethyl phthalate	100000	40000.0	5000.00

[a] Target concentrations were calculated from ADI values using Equation 4 in text.

APPENDIX B

WORKSHOP AGENDA AND INVITED PARTICIPANTS

WORKSHOP ON APPROACHES TO ECOLOGICAL AND HUMAN HEALTH RISK ANALYSIS
FOR DISPOSAL OF CONTAMINATED SEDIMENTS AND HUMAN CONSUMPTION
OF CONTAMINATED SEAFOOD
December 16 and 17, 1985

Conference Room 12A (12th Floor), U.S. EPA Region X Office
1200 Sixth Avenue, Seattle, WA

AGENDA

December 16, 1985

8:30 a.m. INTRODUCTION AND STATEMENT OF OBJECTIVES
Dr. Robert Pastorok - Tetra Tech, Inc.

9:00 a.m. OVERVIEW OF WORKSHOP MANAGEMENT DECISIONS FOR PUGET SOUND

1) PUGET SOUND DREDGED DISPOSAL ANALYSIS
Mr. Keith Phillips - Corps of Engineers; Seattle District

2) PUGET SOUND ESTUARY PROGRAM
Dr. John Armstrong - U.S. EPA, Region X

9:20 a.m. DISCUSSION

9:30 a.m. KEY COMPONENTS OF RISK ASSESSMENT/MANAGEMENT
Dr. Eugene Stakhiv - U.S. Army Engineer Institute for Water Resources
Dr. Curtis Brown - Bureau of Reclamation

9:50 a.m. DISCUSSION

10:00 a.m. KEYNOTE: UPDATE ON RISK ASSESSMENT GUIDELINES AND ISSUES AT U.S. EPA
Dr. Alan Ehrlich - U.S. EPA, Washington, DC

10:30 a.m. DISCUSSION

10:45 a.m. BREAK

11:00 a.m. PITFALLS IN QUANTITATIVE RISK ASSESSMENT
Dr. David Eaton - University of Washington

11:20 a.m. DISCUSSION

11:30 a.m. PRELIMINARY POLLUTANT LIMIT VALUE APPROACH TO EVALUATION OF SOIL/SEDIMENT CONTAMINATION
Dr. David Rosenblatt - U.S. Army Medical Bioengineering R+D Laboratory

11:50 a.m. DISCUSSION

12:00 LUNCH

1:30 p.m. NEW DEVELOPMENTS IN THE DERIVATION AND APPLICATION OF ACCEPTABLE DAILY INTAKE VALUES
Dr. Michael Dourson - U.S. EPA

1:50 p.m. DISCUSSION

2:00 p.m. APPROACHES TO ECOLOGICAL RISK ANALYSIS
Dr. Lawrence Barnthouse - Oak Ridge National Laboratory

2:20 p.m. DISCUSSION

2:30 p.m. RISK ASSESSMENT OF AQUATIC DISPOSAL OF DREDGED MATERIAL
Dr. Jack Gentile - U.S. EPA

2:50 p.m. DISCUSSION

3:00 p.m. BREAK

3:15 p.m. CONCEPTUAL ISSUES IN RISK ASSESSMENT AND RISK MANAGEMENT
Workgroup Discussion of General Conceptual Issues

1) Prioritization of Issues
2) Resolution of Priority Issues
3) Recommendations

5:00 p.m. ADJOURN

December 17, 1985

8:30 a.m. METHODS FOR PREDICTING HUMAN HEALTH RISKS FROM CONSUMPTION OF CHEMICALLY CONTAMINATED SEAFOOD
Workgroup Discussion of Specific Issues

1) Prioritization of Issues
2) Resolution of Priority Issues
3) Recommendation of Risk Assessment Technique(s)

10:00 a.m. BREAK

10:15 a.m. CONTAMINATED SEAFOOD (continued)

11:30 a.m. LUNCH

1:00 p.m. COMPARATIVE RISK ANALYSIS FOR ECOLOGICAL AND HUMAN HEALTH RISKS FROM CONTAMINATED SEDIMENTS IN VARIOUS ENVIRONMENTS
Workgroup Discussion of Specific Issues

1) Prioritization of Issues
2) Resolution of Priority Issues
3) Recommendation of Risk Management Technique(s)

2:00 p.m. BREAK

2:15 p.m. CONTAMINATED SEDIMENTS (continued)

5:00 p.m. ADJOURN

APPENDIX C

CARCINOGENIC POTENCY FACTORS AND REFERENCE DOSE VALUES FOR PRIORITY POLLUTANTS

TABLE C-1. CARCINOGENIC PRIORITY POLLUTANTS RANKED BY POTENCY FACTORS

PP#	Pollutant	CAS Number	Potency[a]	Level of Evidence[b] Humans	Animals
129	TCDD (dioxin)	1746-01-6	156000.00000	I	S
5	benzidine	92-87-5	234.00000 (W)	S	S
119	chromium VI[c]		41.00000 (W)	S	S
90	dieldrin	60-57-1	30.40000	I	S
61	N-nitrosodimethylamine	62-75-9	25.90000 (B)	I	S
115	arsenic		15.00000 (H)	S	I
73	benzo(a)pyrene	50-32-8	11.50000	I	S
89	aldrin	309-00-2	11.40000	I	L
102	alpha-HCH	319-84-6	11.12000	I	S
118	cadmium[c]		6.10000 (W)	L	S
106	PCB-1242	53469-21-9	4.34000	I	S
107	PCB-1254	11097-69-1	4.34000	I	S
108	PCB-1221	11104-28-2	4.34000	I	S
109	PCB-1232	11141-16-5	4.34000	I	S
110	PCB-1248	12672-29-6	4.34000	I	S
111	PCB-1260	11096-82-5	4.34000	I	S
112	PCB-1016	12674-11-2	4.34000	I	S
100	heptachlor	76-44-8	3.37000	I	S
117	beryllium[c]		2.60000	L	S
103	beta-HCH	319-85-7	1.84000	I	L
28	3,3'-dichlorobenzidine	91-94-1	1.69000	I	S
9	hexachlorobenzene	118-74-1	1.67000	I	S
91	chlordane	57-74-9	1.61000	I	L
105	gamma-HCH	58-89-9	1.33000	I	L
29	1,1-dichloroethene	75-35-4	1.16000 (I)	I	L
18	bis(2-chloroethyl)ether	111-44-4	1.14000	I	S
113	toxaphene	8001-35-2	1.13000	I	S
124	nickel (subsulfide, refinery dust)[c]		1.05000 (W)	S	S
37	1,2-diphenylhydrazine	122-66-7	0.77000	I	S
92	4,4'-DDT	50-29-3	0.34000	I	S
93	4,4'-DDE	72-55-9	0.34000	I	S
94	4,4'-DDD	72-54-8	0.34000	I	S
35	2,4-dinitrotoluene	121-14-2	0.31000	I	S
3	acrylonitrile	107-13-1	0.24000 (W)	L	S
15	1,1,2,2-tetrachloroethane	79-34-5	0.20000	I	L
6	tetrachloromethane	56-23-5	0.13000	I	S
10	1,2-dichloroethane	107-06-2	0.09100	I	S
52	hexachlorobutadiene	87-68-3	0.07750	I	L
23	chloroform	67-66-3	0.07000	I	S
14	1,1,2-trichloroethane	79-00-5	0.05730	I	L
85	tetrachloroethene	127-18-4	0.05100	I	L
4	benzene	71-43-2	0.02900 (W)	S	S
21	2,4,6-trichlorophenol	88-06-2	0.01990	I	S
88	vinyl chloride	75-01-4	0.01750 (I)	S	S
12	hexachloroethane	67-72-1	0.01420	I	L
87	trichloroethene	79-01-6	0.01100	I	L/S
62	N-nitrosodiphenylamine	86-30-6	0.00492	I	S
44	dichloromethane	75-09-02	0.00063 (I)	I	L

[a] From U.S. Environmental Protection Agency (1985a), Table 9-66. All slopes calculated as upper 95 percent confidence limit of slope (q_1^*) based on animal oral data and multistage model except:

(B) = slope calculated from 1-Hit model
(W) = slope calculated from occupational exposure
(H) = slope calculated from human drinking water exposure
(I) = slope calculated from animal inhalation studies.

[b] S = Sufficient evidence; L = Limited evidence; I = Inadequate evidence.

[c] Chromium (VI), cadmium, beryllium, and nickel are not considered to be carcinogenic via dietary exposure.

TABLE C-2. REFERENCE DOSE (RfD) VALUES FOR PRIORITY POLLUTANTS

PP#	Pollutant	CAS #	RfD mg/day	RfD mg/kg/day	Criteria Page
126	silver	7440-22-4[a]	0.016	0.0002	C-125
60	4,6-dinitro-o-cresol	534-52-1	0.027	0.0004	C-93
127	thallium	563-68-8[a]	0.04	0.0004	*
56	nitrobenzene	98-95-3	0.03	0.0005	*
42	bis(2-chloroisopropyl)ether	39638-32-9	0.070	0.001	C-61
98	endrin	72-20-8	0.070	0.001	B-12
123	mercury	7439-97-6[a]	0.1	0.002	*
59	2,4-dinitrophenol	51-28-5	0.14	0.002	C-92
33	1,3-dichloropropene	10061-02-6	0.175	0.002	C-27
119	chromium VI	7440-47-3[a]	0.175	0.002	C-34
95	alpha-endosulfan	115-29-7	0.28	0.004	C-87
96	beta-endosulfan	115-29-7	0.28	0.004	C-87
97	endosulfan sulfate	1031-07-8	0.28	0.004	C-87
114	antimony	7440-36-0[a]	0.29	0.004	C-70
39	fluoranthene	206-44-0	0.4	0.006	C-47
53	hexachlorocyclopentadiene	77-47-4	0.418	0.006	C-63
125	selenium	7782-49-2	0.7	0.01	C-66
25	1,2-dichlorobenzene	95-50-1	0.94	0.01	C-64
26	1,3-dichlorobenzene	541-73-1	0.94	0.01	C-64
27	1,4-dichlorobenzene	106-46-7	0.94	0.01	C-64
7	chlorobenzene	108-90-7	1.008	0.01	C-20
2	acrolein	107-82-8	1.100	0.02	C-53
46	bromomethane	74-83-9	1.5	0.02	
124	nickel	7440-02-0[a]	1.5	0.02	
121	cyanide	57-12-5[a]	2	0.02	*
64	pentachlorophenol	87-86-5	2	0.03	*
44	dichloromethane	75-09-02	4	0.06	*
38	ethylbenzene	100-41-4	7	0.1	*
31	2,4-dichlorophenol	120-83-2	7.0	0.1	C-32
65	phenol	108-95-2	7	0.1	*
54	isophorone	78-59-1	10.5	0.150	C-20
86	toluene	108-88-3	20	0.3	*
11	1,1,1-trichloroethane	71-55-6	37.5	0.5	C-77
45	chloromethane	74-87-3	38	0.5	
66	bis(2-ethylhexyl)phthalate	117-81-7	42	0.6	C-57
68	di-n-butyl phthalate	87-74-2	88	1	C-57
119	chromium III	7440-47-3[a]	125	2	
71	dimethyl phthalate	131-11-3	700	10	C-57
70	diethyl phthalate	84-66-2	875	10	C-57

[a] CAS numbers for these substances vary depending on their specific form (e.g., inorganic salts or organic complexes.

Reference: U.S. EPA (1980b). Priority pollutant numbers are shown in first column of table. For each RfD, page citation for corresponding Acceptable Daily Intake value from a Water Quality Criteria document is shown in last column. Blanks in page citation column indicate that RfD values are errata to water quality criteria (U.S. EPA, 8 August 1984, personal communication).

Asterisk indicates that values are verified RfDs from U.S. EPA (1986).

APPENDIX D

RANGE ESTIMATING PROGRAM RESULTS

Range Estimating Program

(Dredge Disposal Demo)

Question: "What is the possible range of human annual PAH consumption from contaminated mussels, given the uncertainty in; (1) the PAH concentrations in disposal material, (2) the rate of uptake in mussels, (3) the rate of human mussel consumption, and (4) the percent of the harvest taken from disposal areas?"

Variables	Units	Target	Lowest	Highest
V1. Dredge PAH	ppm	500	400	2000
V2. Mussel PAH uptake rate	%	50	10	250
V3. Yrly mussel consump/pers	g	2500	250	7500
V4. % of yrly consump. from disposal area	%	15	0	100

Model Specification

1. D1 = % V2*V1 (D1 = ppm PAH in mussels)

2. D2 = D1*V3 (D2 = annual consump. PAH/person if all mussels are taken from disposal area.)

3. D3 = V4*D2 (D3 = micrograms annual PAH consump/person of PAH, adjusted for percent mussels taken from disposal area.)

The Range Estimating Program runs a Monte Carlo simulation using the model specified by the user. One thousand scenarios are created, based on the probability distributions for each input variable (V1 thru V4), resulting in 1000 outcomes (micrograms annual human consumption). This provides a probability distribution of outcomes, as shown on page 2 and 3.

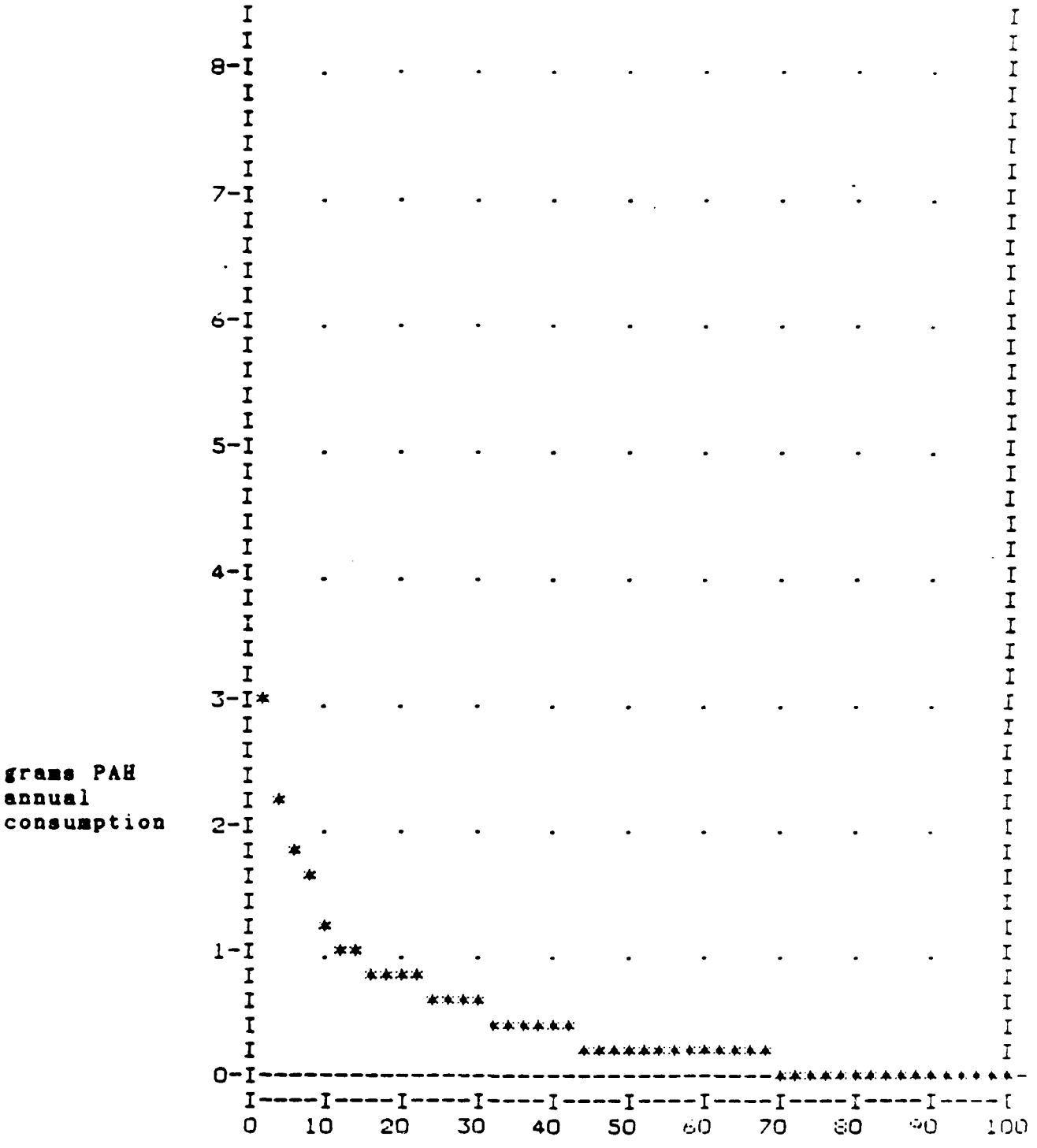
grams PAH
annual
consumption
8
7
6
5
4
3
2
1
0
0
10
20
30
40
50
60
70
80
90
100
Probability of Outcome (percent)

Table 1

Probability of Exceeding Specific Levels of Consumption (micrograms PAH)

9518784--- *	2202701--- 5	1259386---10	931394---1
755776---20	622621---25	510156---30	429667---3
360833---40	284288---45	238770---50	192954---5
149032---60	117469---65	93341---70	76970---7
62834---80	43042---85	28299---90	14081---9
1265---**			

* LESS THAN .10 PERCENT PROBABILITY THE TOTAL WILL EXCEED THIS
** GREATER THAN 99.90 PERCENT PROBABILITY THE TOTAL WILL EXCEED THIS

2 EXAMPLES TO SHOW HOW TO INTERPRET THIS PROFILE

THERE IS A 20 PERCENT PROBABILITY THE TOTAL WILL EXCEED 755776
THERE IS A 80 PERCENT PROBABILITY THE TOTAL WILL EXCEED 62834

APPENDIX E

HYPOTHETICAL DATA FOR EXAMPLE RISK ASSESSMENT

TABLE E-1. HYPOTHETICAL EXAMPLE OF TOTAL OR BULK CONTAMINANT CONCENTRATIONS IN FOUR PUGET SOUND SEDIMENTS

Contaminants of Concern	Sediment			
	Reference	A	B	C
As	5.5	9,700	90.0	14.0
Cd	0.24	184	3.6	1.6
Cu	54.0	11,400	239.0	115.0
Pb	10.0	6,250	181.0	81.0
Hg	0.10	52	0.50	0.18
Zn	50.8	3,320	242.0	107.0
Base/neutrals				
Naphthalene	0.029	0.540	1.012	0.350
Fluorene	0.007	0.835	0.600	0.625
Phenanthrene	0.070	0.760	1.210	0.600
Fluoranthene	0.030	0.870	12.250	1.500
Pyrene	0.065	1.350	8.800	0.150
Benzo(a)pyrene	0.060	1.050	6.190	0.190
Hexachlorobutadiene	0.029	0.025	0.480	0.180
Hexachlorobenzene	0.065	1.280	1.050	0.220
Acid extractable				
Pentachlorophenol	0.030	0.100	0.100	0.350
Pesticides				
PCB (total)	0.025	0.260	2.000	1.245
Sand, percent	30.0	66.7	20.2	38.7
Silt, percent	40.0	25.2	54.7	42.3
Clay, percent	30.0	7.8	25.1	19.0
TOC, percent	2.5	8.8	4.4	2.9

Note: Values in mg/kg dry weight, except as otherwise indicated.

Reference: U.S. Army Corps of Engineers (1985)

TABLE E-2. HYPOTHETICAL EXAMPLE OF CONCENTRATIONS OF DISSOLVED CONTAMINANTS IN THE SATURATED LEACHATE OF A NEARSHORE AREA CONTAINING THREE PUGET SOUND SEDIMENTS

Contaminants of Concern	Chronic Criterion-Saltwater*	Drinking Water Standard**	Reference Water	Sediment		
				A	B	C
As	--†	50	2	560	12	1
Cd	4.5	10	1	0.5	0.01	0.004
Cu	4.0	1,000	17	200	8.2	3.0
Pb	--	50	1.0	0.5	0.02	0.014
Hg	0.025	2	0.1	0.07	0.007	0.003
Zn	58	5,000	10	150	62.5	25
Base/neutrals						
Naphthalene	--	--	<1	<1	120	<1
Fluorene	--	--	<1	<1	10	<1
Phenanthrene	--	--	<1	<1	110	<1
Fluoranthene	--	--	<1	<1	150	12
Pyrene	--	--	<1	<1	11	<1
Benzo(a)pyrene	--	--	<1	<1	11	<1
Hexachlorobutadiene	--	--	<1	<1	10	<1
Hexachlorobenzene	--	--	<1	<1	10	<1
Acid extractable						
Pentachlorophenol	--	--	<1	<1	<1	<1
Pesticides						
PCB (total)	0.03	--	0.05	0.05	0.50	0.25

Note: Values are in µg/ℓ.
* Table C2.
** Table C4.
† --denotes value not established.

Reference: U.S. Army Corps of Engineers (1985)

TABLE E-3. RESULTS OF HYPOTHETICAL AMPHIPOD SEDIMENT BIOASSAY SERIES

Percent Dredged Material	Percent Mortality	Percent Excess Mortality[a]
0 (control)	4	--
0 (control)	10	--
10	10	3
10	15	8
25	25	18
25	15	8
50	60	53
50	65	58
75	85	78
75	60	53
100	90	83
100	75	68

[a] Percent Excess Mortality (PB_r) = Percent mortality in dredged material bioassay minus average percent mortality in reference sediment bioassay = percent mortality minus 7 percent.

END

10-81

DTIC